APPENDIX C

EVALUATION CRITERIA

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Term	Definition
BA	Biological Assessment
BMP	best management practice
BOD	biological oxygen demand
C	Celsius
CDFG	California Department of Fish and Game
CRWQCB	California Regional Water Quality Control Board
cfs	cubic-feet per second
cm	centimeter(s)
cm/hr	centimeters per hour
CVFF	Coyote Valley Fish Facility
d_{50}	median diameter
DCFH	Don Clausen Fish Hatchery (also known as Warm Springs Fish Hatchery)
DO	dissolved oxygen
EPA	U.S. Environmental Protection Agency
Estuary	Russian River Estuary
ESU	Evolutionarily Significant Unit
fps	feet per second
HSI	Habitat Suitability Index
ILT	Incipient Lethal Temperature
JTU	Jackson turbidity unit
mg/l	Milligram(s) per liter
mm	millimeter(s)
MCRRFCD	Mendocino County Russian River Flood Control and Water Conservation Improvement District
MWAT	maximum weekly average temperature
NATURES	Natural Rearing Enhancement System

Term	*Definition*
NCRWQCB	Regional Water Quality Control Board, North Coast Region
NMFS	National Marine Fisheries Service, now known as NOAA Fisheries
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NTU	Nephelometric turbidity unit
RMA	Resource Management Associates
RRWQM	Russian River Water Quality Model
SAR	smolt-to-adult survival
SCWA	Sonoma County Water Agency
UIL_{50}	upper incipient lethal temperature at which 50 percent of experimental organisms die within a given timeframe
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WSE	water surface elevation
YOY	young of the year

C.1.1 INTRODUCTION

This appendix to the Russian River Draft Biological Assessment (BA) presents the evaluation criteria used to analyze the potential for effects to the Russian River populations of coho salmon, steelhead, and Chinook salmon as a result of the proposed project. It describes the models used to predict the effects of some alternative operations and describes additional studies that have been conducted to assess project effects.

ENTRIX, Inc. developed evaluation criteria to assess or scale the effects of U.S. Army Corps of Engineers (USACE), Sonoma County Water Agency (SCWA), and Mendocino County Russian River Flood Control and Water Conservation Improvement District (MCRRFCD) activities in a semi-quantitative way. The criteria provide the basis for an objective evaluation of the various activities and facilities that are included in the BA. They provide the basis for assigning an effect score for such effects as water temperature, habitat, sedimentation, and scour.

Evaluation criteria are based on a review of the available information and initial consultation with appropriate technical resources including staff from USACE, SCWA, National Oceanic and Atmospheric Administration (NOAA) Fisheries, California Department of Fish and Game (CDFG), and others. Preliminary criteria were presented to USACE, SCWA, CDFG, NOAA Fisheries, and the North Coast Regional Water Quality Control Board (NCRWQCB) by ENTRIX, Inc. scientists during a meeting. The criteria, the basis for the criteria, and how they would be applied were explained at this meeting. Comments on the proposed criteria were addressed.

An objective basis for assessing effects to the listed fish and their habitats was identified. Effect assessment is based on a consistent set of evaluation categories and criteria for life-history stages of coho salmon, steelhead, and Chinook salmon and their habitats by reach of the Russian River, Dry Creek, and tributaries. If an effect is common among project components, the same criteria are applied. A consistent set of scoring criteria simplifies comparisons between facilities, locations, and effects. The scoring process is automated for hydrological time series and model outputs. The scoring system is set up on a relative scale of 0 to 5. A score of 0 indicates an effect that could result in population failure or substantial adverse modification to habitat, and a score of 5 indicates no negative effect or a beneficial effect. A separate scoring system is set up for each species and life-history stage. Each species/life-history stage group is evaluated separately. The score is applied only on the months when the species/life-history stage being evaluated would be expected to be present in the river.

The criteria provide a consistent means of evaluation of a specific effect by species and life-history stage, and location, as appropriate. For instance, coho salmon may not use some areas in the watershed for juvenile rearing so there would be no reason to evaluate these areas for an effect on coho salmon as long as the facility or operation component had only localized effects.

C.1.2 PHYSICAL HABITAT-RELATED CRITERIA

Habitat may be affected by a number of factors. These include flow, water temperature, water quality, and sediment, among others. Some habitat factors are related to the configuration of the channel and its characteristics, while others affect the fish regardless of location (e.g., dissolved oxygen [DO]). Streamflow affects fish habitat for spawning/incubation and rearing, as well as fish passage. This section addresses flow-related physical habitats and criteria to evaluate those habitats.

Winzler and Kelly (1978) conducted a systematic survey of existing and potential fish habitat in the mainstem Russian River and Dry Creek. Minimum instream flows under D1610 were determined, in part, based on this work. Information was developed for flow-related effects on spawning and rearing habitat by Winzler and Kelly (1978) and by CDFG (Baracco 1977). Because this information is more than 20 years old and channel conditions have changed in the intervening period, this information was not used in developing evaluation criteria for this BA.

In fall 2001, ENTRIX, Inc. developed the Russian River and Dry Creek Flow-Related Habitat Assessment (Flow Habitat Study, Appendix F) (ENTRIX, Inc. 2003) in consultation with USACE, SCWA, NOAA Fisheries, CDFG, and the California Regional Water Quality Control Board (CRWQCB) (see Appendix F). This study was developed as part of the effort to evaluate flow-related habitat under D1610 and other potential flow regimes, and to develop information regarding how fish habitat changes with flow. ENTRIX, Inc., NOAA Fisheries, USACE, and SCWA developed a semi-quantitative analysis of flow-related habitat needs. Study objectives centered on the management of rearing habitat because the study participants believe fry and juvenile rearing may limit fish production in the study area. Habitat quality and quantity were determined by considering a combination of field measurements at representative cross-sectional transects and observations, and qualitative analysis of the available habitat at different evaluation flows by a team of professional fishery scientists from the participating entities above.

The Flow Habitat Study evaluated habitat availability at alternative flows scenarios for juvenile and fry life-history stages of the three listed species of anadromous salmonids in Dry Creek and the Russian River. In addition, spawning habitat for steelhead and Chinook salmon was evaluated for the Russian River, but not for Dry Creek. The study area included Dry Creek between Warm Springs Dam and the Russian River confluence, and the Russian River between the Forks and the City of Cloverdale. Habitat was evaluated over a range of releases from Warm Springs Dam and Coyote Valley Dam. Russian River sites were evaluated during stable dam releases of 125 cfs, 190 cfs, and 275 cfs. Dry Creek sites were evaluated during stable dam release flows of 47 cfs, 90 cfs, and 130 cfs.

Generally speaking, the lower flow levels evaluated provided greater amounts of suitable and optimal rearing habitat than the higher flow levels. On Dry Creek this was particularly true for steelhead fry and juveniles. The low and intermediate flow levels on Dry Creek provided similar amounts of habitat for fry and juvenile Chinook salmon. The amount of habitat at the 130 cfs flow level on Dry Creek provided much less suitable and optimal habitat for both species than either of the two lower flows. In most Dry Creek study sites, at least 25 percent of the stream area provided optimal habitat for steelhead fry and juveniles when flows were either 47 cfs or 90 cfs; most of these cases occurred at the lowest flow. Dry Creek also provided ample nursery habitat for Chinook salmon; at least 25 percent of the stream area was rated optimal at flows of 47 cfs and 90 cfs.

On the Russian River, the lowest observed flow provided the greatest amount of habitat for both Chinook salmon and steelhead fry and juveniles. The intermediate flow provided the greatest amount of habitat for spawners of both species. On the Russian River the difference in the amount of habitat was more similar among the three flow levels and there was not the tremendous decrease in habitat at the highest flow level as was observed in Dry Creek.

Flow criteria were developed based in part on the Flow Habitat Study, described above, as well as conversations with biologists familiar with the Russian River, and professional judgment. Flow evaluation criteria for the effects of instream flow on habitat availability are presented for each species and each life-history stage (Tables C-1 and C-2). These criteria were applied in an analysis of flow in Dry Creek and the mainstem Russian River.

To assess flow-related habitat, flow criteria will be applied to predicted flows from the Russian River System Model (RRSM) under various flow scenarios. This model was developed by SCWA. Flows were modeled at various locations in the Russian River and Dry Creek.

Table C-1 Flow Evaluation Criteria for the Russian River by Species and Life-History Stage

Coho Salmon	**Nov 1 to Jan 31**
Habitat Score	**Q (cfs) Upmigration**
0	≤ 50
1	> 50 ≤ 75
2	> 75 ≤ 100
3	>100 ≤ 125
4	> 125 ≤180
5	> 180 ≤ 400
4	> 400 ≤ 800
3	> 800 ≤ 2000
2	> 2000 ≤ 4000
1	> 4000
0	

Table C-1 Flow Evaluation Criteria for the Russian River by Species and Life-History Stage (Continued)

Coho Salmon Habitat Score	Nov 1 to Jan 31 Q (cfs) Upmigration			
Chinook Salmon **Habitat Score**	**Aug 15 to Jan 15** **Q (cfs)** **Upmigration**	**Nov 1 to Jan 31** **Q (cfs)** **Spawning**	**Feb 1 to Apr 30** **Q (cfs)** **Fry Rearing**	**Apr 1 to Jun 30** **Q (cfs)** **Juvenile Rearing**
0	≤ 50	≤ 25	≤ 0	≤ 0
1	> 50 ≤ 75	> 25 ≤ 100	> 0 ≤ 20	> 0 ≤ 20
2	> 75 ≤ 100	> 100 ≤ 130	> 20 ≤ 40	> 20 ≤ 50
3	>100 ≤ 125	> 130 ≤ 150	> 40 ≤ 80	> 50 ≤ 100
4	> 125 ≤180	> 150 ≤ 190	> 80 ≤ 115	> 100 ≤115
5	> 180 ≤ 400	> 190 ≤ 210	> 115 ≤ 135	> 115 ≤ 145
4	> 400 ≤ 800	> 210 ≤ 300	> 135 ≤ 175	> 145 ≤ 190
3	> 800 ≤ 2000	> 300 ≤ 400	> 175 ≤ 250	> 190 ≤ 275
2	> 2000 ≤ 4000	> 400 ≤ 700	> 250 ≤ 500	> 275 ≤ 1000
1	> 4000	> 700 ≤ 2500	> 500 ≤ 1500	> 1000 ≤ 2500
0		> 2500	> 1500	> 2500
Steelhead **Habitat Score**	**Jan 1 to Mar 31** **Q (cfs)** **Upmigration**	**Jan 1 to Apr 30** **Q (cfs)** **Spawning**	**Mar 1 to Jun 30** **Q (cfs)** **Fry Rearing**	**Summer** **Q (cfs)** **Juvenile Rearing**
0	≤ 50	≤ 25	≤ 0	≤ 0
1	> 50 ≤ 75	> 25 ≤ 70	> 0 ≤ 20	> 0 ≤ 20
2	> 75 ≤ 100	> 70 ≤ 100	> 20 ≤ 40	> 20 ≤ 50
3	>100 ≤ 125	> 100 ≤ 130	> 40 ≤ 80	> 50 ≤ 80
4	> 125 ≤180	> 130 ≤ 180	> 80 ≤ 100	> 80 ≤ 115
5	> 180 ≤ 400	> 180 ≤ 200	> 100 ≤ 125	> 115 ≤ 145
4	> 400 ≤ 800	> 200 ≤ 250	> 125 ≤ 150	> 145 ≤ 190
3	> 800 ≤ 2000	> 250 ≤ 350	> 150 ≤ 200	> 190 ≤ 275
2	> 2000 ≤ 4000	> 350 ≤ 700	> 200 ≤ 500	> 275 ≤ 1000
1	> 4000	> 700 ≤ 2500	> 500 ≤ 1500	> 1000 ≤ 2500
0		> 2500	> 1500	> 2500

Table C-2 Flow Evaluation Criteria for Dry Creek by Species and Life-History Stage

Coho Salmon Habitat Score[1]	Nov 1 to Jan 31 Q (cfs) Upmigration	Dec 1 to Feb 15 Q (cfs) Spawning	Feb 1 to Apr 30 Q (cfs) Fry Rearing	Summer Q (cfs) Juvenile Rearing
0	≤ 10	≤ 5	≤ 0	≤ 0
1	>10 ≤ 20	> 5 ≤ 20	> 0 ≤ 10	> 0 ≤ 10
2	> 20 ≤ 30	> 20 ≤ 30	> 10 ≤ 20	> 10 ≤ 25
3	> 30 ≤ 90	> 30 ≤ 45	> 20 ≤ 30	> 25 ≤ 45
4	> 90 ≤ 125	> 45 ≤ 60	> 30 ≤ 40	> 45 ≤ 60
5	> 125 ≤ 200	> 60 ≤ 80	> 40 ≤ 70	> 60 ≤ 85
4	> 200 ≤ 250	> 80 ≤ 100	> 70 ≤ 90	> 85 ≤ 100
3	> 250 ≤ 325	> 100 ≤ 125	> 90 ≤ 130	> 100 ≤ 120
2	> 325 ≤ 400	> 125 ≤ 250	> 130 ≤ 200	> 120 ≤ 200
1	> 400 ≤ 500	> 250 ≤ 800	> 200 ≤ 500	> 200 ≤ 500
0	> 500	> 800	> 500	> 500
Chinook Salmon **Habitat Score[1]**	**Aug 15 to Jan 15** **Q (cfs)** **Upmigration**	**Nov 1 to Jan 31** **Q (cfs)** **Spawning**	**Feb 1 to Apr 30** **Q (cfs)** **Fry Rearing**	**Apr 1 to Jun 30** **Q (cfs)** **Juvenile Rearing**
0	≤ 10	≤ 5	≤ 0	≤ 0
1		> 5 ≤ 25	> 0 ≤ 10	> 0 ≤ 10
2	> 10 ≤ 45	> 25 ≤ 40	> 10 ≤ 20	> 10 ≤ 25
3	> 45 ≤ 60	> 40 ≤ 60	> 20 ≤ 30	> 25 ≤ 45
4	> 60 ≤ 90	> 60 ≤ 80	> 30 ≤ 45	> 45 ≤ 60
5	> 90 ≤ 125	> 80 ≤ 105	> 45 ≤ 60	> 60 ≤ 90
4	> 125 ≤ 200	> 105 ≤ 130	> 60 ≤ 90	> 90 ≤ 100
3	> 200 ≤ 325	> 130 ≤ 150	> 90 ≤ 110	> 100 ≤ 110
2	> 325 ≤ 400	> 150 ≤ 250	> 110 ≤ 150	> 110 ≤ 200
1	> 400 ≤ 500	> 250 ≤ 1300	> 150 ≤ 500	> 200 ≤ 500
0	> 500	> 1300	> 500	> 500
Steelhead **Habitat Score[1]**	**Jan 1 to Mar 31** **Q (cfs)** **Upmigration**	**Jan 1 to Apr 30** **Q (cfs)** **Spawning**	**Mar 1 to Jun 30** **Q (cfs)** **Fry Rearing**	**Summer** **Q (cfs)** **Juvenile Rearing**
0	≤ 10	≤ 5	≤ 0	≤ 0
1	>10 ≤ 20	> 5 ≤ 20	> 0 ≤ 5	> 0 ≤ 5
2	> 20 ≤ 30	> 20 ≤ 30	> 5 ≤ 15	> 5 ≤ 15
3	> 30 ≤ 90	> 30 ≤ 60	> 14 ≤ 30	> 14 ≤ 30
4	> 90 ≤ 125	> 60 ≤ 80	> 30 ≤ 40	> 30 ≤ 40
5	> 125 ≤ 200	> 80 ≤ 110	> 40 ≤ 55	> 40 ≤ 55
4	> 200 ≤ 250	>110 ≤ 135	> 55 ≤ 70	> 55 ≤ 70
3	> 250 ≤ 325	> 135 ≤ 150	> 70 ≤ 90	> 70 ≤ 90
2	> 325 ≤ 400	> 150 ≤ 250	> 90 ≤ 110	> 90 ≤ 110
1	> 400 ≤ 500	> 250 ≤ 1300	> 110 ≤ 500	> 110 ≤ 500
0	> 500	> 1300	> 500	> 500

[1] A score of “5” is the highest, “1” is the lowest.

Because of the different channel dimensions and channel morphology, separate flow criteria were developed for the Russian River and Dry Creek. In the following, the criteria for the Russian River are discussed first, followed by the criteria for Dry Creek

Russian River

Upstream Migration

Upstream migration was not assessed during the Flow Habitat Study. Upstream migration can be affected when flows so low that depths in riffle habitats are too shallow for fish to swim upstream because they cannot submerge their bodies completely, or when velocities become so high that fish cannot move upstream against these velocities. Conversations with biologists knowledgeable about the Russian River watershed indicated that, in the Russian River, the optimal flows for upstream passage of salmonids would range from about 150 to 400 cfs. In addition, at 100 cfs the fish would likely not be able to submerge completely, but may still be able to move upstream through the riffle. At the high flows, 4,000 cfs was thought to be a flow that would produce velocities sufficient to severely impede migration because of high velocities. However, even at this flow, it was recognized that there would probably still be areas where velocities were low enough for fish to find ways to move upstream.

Based on this information, flows between 180 and 400 cfs were considered optimal for migration, and received a score of 5. A flow of 50 was considered completely impassible because of shallow water depths and received a score of 0. A score of 0 was not assigned at high flows for the reason given above, but all flows over 4,000 cfs were assigned a score of 1, indicating that passage would be severely impaired. Between the optimal range of flows and the unsuitable flows identified above, scores were assigned to describe relative degrees of passage impairment. The flow ranges assigned to each score category are provided in Table C-1.

Spawning

Spawning criteria were developed for steelhead and Chinook salmon on the Russian River. Criteria were not developed for coho salmon as they are not thought to spawn in the Russian River. Spawning habitat for steelhead and Chinook salmon was assessed in the Russian River during the Flow Habitat Study. The Flow Assessment team concluded that the middle flow release of 190 cfs provided the best quality and highest quantity of habitat for spawning for both species. The lowest flow (125 cfs release) resulted in the better spawning habitat than the highest flow release for steelhead, while the reverse was true for Chinook salmon. The decrease in habitat quantity and quality was generally not great compared to the amount available at the middle flow release.

Based on the results of the Flow Habitat Study and professional judgment, flow scoring criteria for the Russian River considered flows of 190 to 210 cfs optimal (score of 5) for Chinook salmon spawning and 180 to 200 as optimal for steelhead spawning. Spawning habitat decreases as flows move away from this range. As flows become lower, depths may become too shallow and velocities too low for spawning. As flows exceed the optimal range, depths and velocities become too great for spawning. Flows of 25 cfs or less were considered too low for successful spawning for both species on the Russian River and given a score of 0. Flows of more than 2,500 cfs would result in velocities that were generally too high for spawning of both species (as well as depths that were too deep) and also given a score of 0. Between the flows for optimal and completely unsuitable habitat (score of 0) ranges of flow were assigned scores based on their relative suitability for spawning. These flow ranges are provided in Table C-1.

Rearing Habitat

Rearing habitat was assessed for fry and juvenile steelhead and Chinook salmon on the Russian River during the Flow Habitat Study. Coho salmon do not rear in the mainstem Russian River and therefore habitat was not assessed nor were criteria developed for coho salmon rearing lifestages. This study found that the lowest flow (125 cfs) provided the best conditions for both of these rearing lifestages for both species. Rearing habitat value decreased gradually as flows increased, and was substantially reduced at the highest study flow (275 cfs release flow) relative to the lowest flow level. For these lifestages, habitat would continue to be available in some areas even at very low flows. However, as flow levels decrease, these fish would be crowded into ever smaller areas, forced into pools with potential predators, and be exposed to warmer water temperatures and decreased water quality. Habitat would continue to be available, however, for some fish. Based on this, a flow of 0 was assigned a score of 0.

As flows increase above the optimal range, velocities begin to increase to levels higher than optimal. Juveniles and fry are forced to seek out areas of reduced velocity. As flows continue to increase, these areas become smaller and smaller, crowding this fish into smaller areas of suitable habitat. At flows of 1,500 cfs for fry and 2,500 cfs for juveniles, velocities become high enough that there is little area with suitable velocities left for fry and juveniles to live in. These flows were assigned a score of 0 for the two lifestages. Between the range of optimal flows and the flows that provide little or no habitat, scores were assigned that reflect the relative value of different flow ranges as rearing habitat. These scores are provided in Table C-1.

<u>Dry Creek</u>

The flow ranges providing habitat for the various lifestages in Dry Creek are lower than those in the Russian River because of the smaller size of the channel. However, the scoring criteria were derived in the same manner as those for the Russian River. The Flow Habitat Study evaluated fry and juvenile rearing habitat for all three anadromous salmonid species in Dry Creek (coho salmon, steelhead, and Chinook salmon). Habitat conditions were not evaluated for spawning as part of this study. For both spawning and upstream migration, the scoring criteria are based upon discussions with biologists knowledgeable about the Russian River watershed and professional judgment.

Upstream Migration

For upstream migration, flows of 90 to 125 cfs were considered optimal for upstream migration. Flows of 10 cfs or less were considered to provide depths too shallow for passage over riffles, and flows of more than 500 cfs were thought to provide velocities too high for fish to migrate upstream. Between the optimal range and the flows that prevented passage, scores were assigned to describe the relative value of different flow ranges for passage. Flows between 30 and 325 cfs were thought to provide acceptable conditions for passage and were scored 3 or higher. As flows declined below 30 cfs, depths become too shallow for passage, and passage becomes increasingly difficult the more flows decline. At flows of 325 cfs there are likely areas in the riffles where velocities are still passable and most fish would be able to migrate upstream. As flows increase further, velocities become higher and passage becomes increasingly more difficult. The range of flows within each score category for upstream migration are provided in Table C-2.

Spawning

Spawning habitat was considered to be optimal at flows from 60 to 80 cfs for coho salmon, 80 to 105 cfs for Chinook salmon, and 80 to 110 cfs for steelhead. Flows of 5 cfs were considered to provide no suitable spawning habitat for any of the species and given a score of 0. Velocities were considered to become unacceptably high for spawning at 800 cfs for coho salmon and 1,300 cfs for Chinook salmon and steelhead. These flows were given a score of 0 for the respective species. The range of flows resulting in intermediate scores is provided in Table C-2.

Rearing

Fry and juvenile rearing habitat was assessed at flows of 47, 90, and 130 cfs as part of the Flow-Habitat Study. This study found that the lowest flow level provided the best habitat (greatest quantity and best quality) for both juvenile and fry of all three species. At 90 cfs, habitat was somewhat worse than that at the lowest flow level, but at the 130 cfs release, habitat was severely reduced. Fry habitat was considered optimal at flows ranging from 40 to 70 cfs for coho salmon, 45 to 60 cfs for Chinook salmon, and 40 to 55 cfs for steelhead. Flows of 0 cfs were given a score of 0. Low flows may continue to provide habitat in pools; however, water quality and temperature conditions may deteriorate at extremely low flows. Additionally, food-producing areas (riffles and runs) would be reduced in size. As flows increase above the optimal range of flows, high velocities become an increasing problem. At 500 cfs, velocities are too high for fry and juvenile rearing for all three species and flows of 500 or more are scored a 0. The flow ranges assigned to intermediate score values for salmonid rearing are described in Table C-2.

C.1.3 WATER QUALITY CRITERIA

Water quality parameters include temperature, DO, and turbidity.

C.1.3.1 TEMPERATURE

Water temperature directly affects an organism's ability to survive, grow, and reproduce. It is one of the most important factors controlling the production and distribution of freshwater fish. In general, the growth rate and physiological performance (e.g., swimming ability) of salmonids increases as a function of temperature up to some tolerance level and then declines as temperatures continue to climb. Excessively high, sustained temperatures reduce growth and physiological performance, increase susceptibility to disease, and may ultimately result in death. Factors such as DO levels and food availability affect temperature tolerance of salmonids.

Published temperature requirements for salmonids are characterized as preferred, optimum, or tolerable. The term "preferred" refers to the temperature range fish species most frequently occupy when placed in a thermal gradient, and are often considered a reasonable estimator of beneficial/optimal temperatures. The optimum temperature range is that at which feeding activity and physiological response is most efficient (McCullough 1999). Tolerable temperature ranges are those in which salmonids can survive.

To determine upper lethal temperatures, two basic methods are used in studies: 1) critical thermal maximum (CTM) and 2) incipient lethal temperature (ILT). The CTM method slowly heats water to find the upper tolerance levels for fish, while the ILT method abruptly transfers fish to warmer water. Temperature thresholds for tolerable ranges are determined experimentally by placing fish in different water temperatures and measuring mortality rates.

The temperatures that may be considered deleterious for a fish species depend on the duration of exposure. The US Environmental Protection Agency (EPA 2001) cites National Academy of Sciences (NAS) (1972) recommendations for water temperature exposure for protection of aquatic life that specify maximum acceptable temperatures for prolonged exposures (>1 week), winter maximum temperatures, short-term exposure to extreme temperature, and suitable reproduction and development temperatures. Lethal effects are thermal effects that cause direct mortality within an exposure period of less than 1 week. Survival rates based on amount of time exposed and temperature of exposure are extremely well described in the scientific literature. The upper incipient lethal temperature (UILT) is an exposure temperature, given a previous acclimation to a constant acclimation temperature, that 50 percent of the fish can tolerate for seven days (Elliott 1981). Alternatively, UILT at a particular acclimation temperature has been determined as an exposure temperature producing 50 percent survival within 1,000 minutes (Brett 1952, Elliott 1981) or 24 hours (Wedemeyer and McLeay 1981, Armour 1990). For salmonids, a survey of the literature indicates that acclimation temperatures above approximately 68°F (20°C) produce similar UILT values, although very small increases in UILT can occur at up to a 75.2°F (24°C) acclimation temperature. Consequently, it can be safely assumed that any UILT study in which acclimation temperature was 68°F (20°C) will produce a UILT nearly identical to the UUILT (ultimate UILT). UILTs reported by EPA (2001) for rainbow trout range from 24° to 26.9°C.

While these experimental values are useful for assessing temperature requirements, they do not take into account salmonid adaptations to regional temperature regimes. Furthermore, salmonids can withstand short-term exposure to temperatures higher than those needed for longer term growth or survival without significant negative effects.

Optimal, tolerable, and lethal water temperature ranges vary by species and by life-history stage (e.g., salmonid embryos are less tolerant of high temperatures than juveniles). For instance, adult salmon and steelhead may delay upstream spawning migrations if water temperatures are too warm or too cold. In cool temperatures, gamete maturation is slowed, which can lead to delays in spawning. Although salmonids have some natural flexibility in migration schedules, human-induced changes in temperature regime may produce unfavorable conditions for native stocks to persist and drive them to extinction if temperature changes become too extreme.

Embryo development is also sensitive to extremes in water temperature. In general, there are high and low threshold values beyond which embryo mortality increases. Embryos can survive at temperatures near lower threshold values, provided initial development of the embryo has progressed to a stage that is tolerant of cold water.

Finally, water temperature has a large impact on the metabolism (since salmonids are cold-blooded) and growth of juvenile salmonids. While there is an optimum temperature range at which growth is maximized, most salmonid species can grow fairly quickly at higher than optimal temperatures, given an adequate food supply. If food is not plentiful enough to meet elevated metabolic needs of the fish in higher water temperatures, fish could experience a slower growth rate or weight loss.

There are no regional or Russian River-specific temperature data to assess the effects of temperature on coho salmon, steelhead, and Chinook salmon. Stream temperatures that restrict salmonids vary with species and by geographic region. Critical temperatures that limit production and survival also vary widely in the literature.

The NCRWQCB reviewed the water quality objective for temperature in the Russian River basin to protect aquatic life, including listed species. This process included an in-depth analysis of salmonid water temperature tolerances. The NCRWQCB staff report (NCRWQCB 2000) identified maximum temperatures and preferred temperature ranges for each species and lifestage. The report identified five alternatives for the revision of the water quality objective for temperature. These alternatives are undergoing further review, and a final selection has not been made.

The following sections document preferred water temperatures and thermal tolerance ranges for coho salmon, steelhead, and Chinook salmon. These values are used to develop evaluation criteria for the effects of temperature for four different life-history stages: upstream migration, adult spawning, incubation of embryos, and rearing of juveniles. Literature values and citations used to develop temperature evaluation criteria are listed in Attachment 1.

C.1.3.1.1 Temperature Criteria

To quantify the effect of temperature change on salmonid persistence, evaluation criteria that assign scores to various temperature ranges were developed. The temperature criteria are based on the peer-reviewed literature values discussed below and are consistent with recently-conducted temperature reviews (NCRWQCB 2000, Myrick and Cech 2001, Sullivan et al. 2000). In developing the criteria for the Russian River, literature values based on California stocks were given preference.

The effect of temperature on a given life-history stage is quantified using scores from 0 to 5. Preferred temperatures are based on optimum ranges from the literature and are assigned a score of 5. ILTs are given a score of 0, despite the fact that these are the lowest magnitude temperatures that can result in mortality (i.e., upper ILT at which 50 percent of experimental organisms die within a given time-frame [UIL_{50}]). The distribution of scores between 0 and 5 is based either on literature values, or in cases where no literature values exist, is interpolated between known values (see Tables A to D in Attachment 1). The distribution of criteria values is based on U.S. Fish and Wildlife Service (USFWS) Habitat Suitability Index (HSI) curves to the extent feasible.

Most of the literature values used to develop the criteria are based on studies conducted in the Pacific Northwest, and may not reflect upper temperature limits of salmonids in the southern portion of their range. Salmonids in the warmer portion of their range may have local adaptations to their regional temperature. For example, steelhead can survive in higher summer temperatures if food is plentiful enough to support a higher metabolic rate (Smith and Li 1983). If primary and secondary production is high, then a numeric temperature objective specific to the Russian River may be higher than research based on colder climates would indicate. However, if food production is insufficient, higher temperatures could be detrimental (NCRWQCB 2000).

The temperature criteria for coho salmon, steelhead, and Chinook salmon are presented in Table C-3. To assess project effects on salmonid habitat, the criteria scores were applied to temperature values predicted by the RRWQM (RMA 1995) for various flow regimes. The RRWQM predicts temperature and DO concentrations at nodes located along the Russian River and Dry Creek. Two daily periods are predicted, one at 6:00 a.m. and the other at 6:00 p.m. for each day from January 1, 1929 to September 30, 1995. The mean of these values is used to represent the average daily temperature and DO.

It is recognized that salmonids can withstand short-term exposure to temperatures higher than those required on average without significant negative effects. Because the ILT_{50} values reviewed in the literature and used in the evaluation criteria are based on a 96-hour exposure, an average score is calculated over a 4-day period using a model value for each 24-hour period.

Table C-3 Temperature Evaluation Criteria by Species and Life-History Stage

Coho Salmon

Score[1]	Nov 1 to Jan 31 Upmigration[2]		Dec 1 to Feb 15 Spawning		Dec 1 to Mar 31 Incubation		Oct 1 to Sept 30 Rearing	
0		≤ 3.0		≤ 1.7		≤ 0.0		≤ 1.7
1	> 3.0	≤ 4.0	> 1.7	≤ 3.0	> 0.0	≤ 3.0	> 1.7	≤ 4.0
2	> 4.0	≤ 5.0	> 3.0	≤ 4.0	> 3.0	≤ 3.5	> 4.0	≤ 7.0
3	> 5.0	≤ 6.0	> 4.0	≤ 6.0	> 3.5	≤ 4.0	> 7.0	≤ 8.0
4	> 6.0	< 7.2	> 6.0	< 7.0	> 4.0	< 4.4	> 8.0	< 12.0
5	≥ 7.2	≤ 12.7	≥ 7.0	≤ 13.0	≥ 4.4	≤ 13.3	≥ 12.0	≤ 14.0
4	> 12.7	≤ 14.0	> 13.0	≤ 14.0	> 13.3	≤ 14.0	> 14.0	≤ 15.0
3	> 14.0	≤ 15.0	> 14.0	≤ 15.0	> 14.0	≤ 15.0	> 15.0	≤ 16.0
2	> 15.0	≤ 16.0	> 15.0	≤ 16.0	> 15.0	≤ 16.0	> 16.0	≤ 20.0
1	> 16.0	< 21.1	> 16.0	< 17.0	> 16.0	< 18.0	> 20.0	< 26.0
0	≥ 21.1		≥ 17.0		≥ 18.0		≥ 26.0	

Steelhead

Score	Nov 1 to Jan 31 Upmigration		Dec 1 to Feb 15 Spawning		Dec 1 to Mar 31 Incubation		Oct 1 to Sept 30 Rearing	
0		≤ 4.0		≤ 4.0		≤ 1.5		≤ 0.0
1	> 4.0	≤ 5.0	> 4.0	≤ 5.0	> 1.5	≤ 3.0	> 0.0	≤ 2.0
2	> 5.0	≤ 6.0	> 5.0	≤ 6.0	> 3.0	≤ 4.5	> 2.0	≤ 4.0
3	> 6.0	≤ 7.0	> 6.0	≤ 7.0	> 4.5	≤ 6.0	> 4.0	≤ 8.0
4	> 7.0	< 7.8	> 7.0	< 7.8	> 6.0	< 7.8	> 8.0	< 12.8
5	≥ 7.8	≤ 11.0	≥ 7.8	≤ 11.1	≥ 7.8	≤ 11.1	≥ 12.8	≤ 15.6
4	> 11.0	≤ 13.0	> 11.1	≤ 14.0	> 11.1	≤ 13.0	> 15.6	≤ 18.0
3	> 13.0	≤ 15.0	> 14.0	≤ 16.0	> 13.0	≤ 15.0	> 18.0	≤ 20.0
2	> 15.0	≤ 17.0	> 16.0	≤ 18.0	> 15.0	≤ 17.0	> 20.0	≤ 23.9
1	> 17.0	< 21.1	> 18.0	< 20.0	> 17.0	< 20.0	> 23.9	< 26.0
0	≥ 21.1		≥ 20.0		≥ 20.0		≥ 26.0	

Chinook Salmon

Score	Nov 1 to Jan 31 Upmigration		Dec 1 to Feb 15 Spawning		Dec 1 to Mar 31 Incubation		Oct 1 to Sept 30 Rearing	
0		≤ 0.8		≤ 1.0		≤ 1.0		≤ 1.0
1	> 0.8	≤ 3.0	> 1.0	≤ 2.5	> 1.0	≤ 2.0	> 1.0	≤ 4.0
2	> 3.0	≤ 5.2	> 2.5	≤ 3.5	> 2.0	≤ 3.0	> 4.0	≤ 6.0
3	> 5.2	≤ 7.9	> 3.5	≤ 4.5	> 3.0	≤ 4.0	> 6.0	≤ 8.0
4	> 7.9	< 10.6	> 4.5	< 5.6	> 4.0	< 5.0	> 8.0	< 12.0
5	≥ 10.6	≤ 15.6	≥ 5.6	≤ 13.9	≥ 5.0	≤ 12.8	≥ 12.0	≤ 14.0
4	> 15.6	≤ 17.0	> 13.9	≤ 14.5	> 12.8	≤ 14.2	> 14.0	≤ 17.0
3	> 17.0	≤ 18.4	> 14.5	≤ 15.2	> 14.2	≤ 15.0	> 17.0	≤ 20.0
2	> 18.4	≤ 19.8	> 15.2	≤ 16.0	> 15.0	≤ 15.8	> 20.0	≤ 23.0
1	> 19.8	< 21.1	> 16.0	< 16.7	> 15.8	< 16.7	> 23.0	< 26.0
0	≥ 21.1		≥ 16.7		≥ 16.7		≥ 26.0	

1 Low scores are associated with water temperatures that are too low and too high.
2 Water temperature in °C.
References: Anonymous 1971; Bell 1986; Bjornn and Reiser 1991; Boles et al. 1988; Brett 1952; Brett et al. 1982; CDFG 1991; Resources Agency 1989; Cramer 1992; Fryer and Pilcher 1974; Hallock et al. 1970; Hanel 1971; McMahon 1983; Myrick and Cech 2000, 2002a, 2002b; Raleigh et al. 1984; Rich 1987; Seymour 1956; and EPA 1974.

C.1.3.1.2 Coho Salmon Temperature Requirements

Water temperature can affect upstream migration of coho salmon through direct mortality, increased susceptibility to disease, and delays in migration timing and maturation rates (Holt et al. 1975). Temperatures greater than 25.5°C are lethal to migrating adults (Bell 1973, cited in McMahon 1983), while prolonged exposure to sublethal temperatures can result in major prespawning mortality. Coho salmon are reported to migrate upstream at water temperatures between 7.2°C and 15.6°C (Bell 1991). However, as infection rates in coho salmon can increase dramatically above 12.7°C, spawning temperatures $\leq$ 13°C are recommended to minimize prespawning mortality (McMahon 1983). Based on these literature values, a score of 5 is assigned to water temperatures between 7.2ºC to 12.7ºC and a score of 4 for temperatures of 12.7ºC to 14.0ºC (Table C-3) (see Table A in Attachment 1). The score was decreased for every one degree increase in temperature, except for temperature range of 16°C to 21.1ºC which was assigned a score of 1. Temperatures greater than 21.1ºC are potentially sublethal and were assigned a score of 0 (McMahon 1983, using Hallock 1979 as an estimator). Scores for temperatures below 7.2ºC decline from 4 to 0 in one degree increments. Temperatures below 3.0ºC are assumed to severely affect migration.

Burner (1951, cited by McMahon 1983) observed coho salmon to spawn at temperatures between 2.8°C and 12.2°C. Temperatures in the range of 3°C to 7°C, however, are believed to slow maturation rates and thus reduce survival (Reingold 1968, in Bjornn and Reiser 1991). An EPA report (EPA 1974) for coho salmon on the Columbia River reported a preferred spawning temperature range of 7°C to 13°C, and that mortality rates increase when these threshold values are exceeded. Bell (1986) reports a narrower preferred range of 10°C to 12.8°C. Based on these literature values, a score of 5 (which represents optimal temperatures) is given to temperatures of 7.0°C to 13.0ºC during the spawning season (EPA 1974), and scores of 4 and 3 (which represent acceptable temperature ranges) to temperatures from 13.0°C to 15.0ºC (McMahon 1983). Scores of 1 and 0 are given to temperatures above 16ºC or below 3ºC (McMahon 1983, Bjornn and Reiser 1991).

Incubation temperatures for salmonid species affect embryo mortality, the time to embryo hatching and the physiology of emerging fry. Embryo viability is sensitive to high water temperature values, with elevated levels of mortality occurring when temperatures exceed 14°C (Reiser and Bjornn 1979). Survival rates of emerging fry increase as the duration of the incubation period decreases, which is inversely correlated with temperature (i.e., higher temperatures give rise to shorter incubation times).

Bell (1986) reports a preferred incubation water temperature of 10°C to 12.8°C, based on the length of incubation time and optimal growth rate of emerging fry. Stein et al. (1972) found that growth rates of coho salmon fry were highest for temperatures between 8.9°C and 12.8°C, but growth rates decreased at 18.1°C. Bjornn and Reiser (1991) suggest the optimal water temperatures for coho salmon incubation have a larger range of 4.4°C to 13.4ºC and that embryo survival only decreases once these values are exceeded. McMahon (1983) reports that coho salmon stop growing altogether at temperatures above

20.3°C, and mortality rates increase as a function of temperature above 12°C. Based on these literature values, a score of 5 is assigned to temperatures between 4.4°C to 13.3ºC during the incubation period (Bjornn and Reiser 1991, Bell 1986). Scores of 4 and 3 are given to water temperatures between 13.3°C to 15.0ºC, and scores of 2 and 1 to temperatures between 15.0°C to 18.0ºC (McMahon 1983). A score of 0 is given to temperatures above 18.0ºC, representing an unacceptable level of mortality. Scores for temperatures below 4.4ºC decrease by one unit for every 0.5ºC decrease in temperature. Water temperatures less than 3°C represent the lower lethal limit for embryo survival and are given a score of 1.

Juvenile coho salmon rear at temperatures between 3.3°C and 20.6°C (Bell 1991). Brett (1952) experimentally determined a lower ILT for coho salmon of 0.8ºC and an upper ILT of 26.2ºC. Preferred water temperatures are reported to range from 11.7°C to 14.4°C (Bell 1991), and 12°C to 14°C (Brett 1952). Growth of juveniles is slowed at low temperatures due to a reduction in metabolic processes and at high temperatures because the amount of food required for physiological maintenance increases (Bjornn and Reiser 1991). Based on these literature values, a score of 5 is given to temperatures between 12.0°C to 14.0ºC for the coho salmon rearing period (Bjornn and Reiser 1991, Brett 1952). Scores of 4 and 3 are given to temperatures between 14.0°C and 16.0ºC (McMahon 1983) and scores of 2 and 1 to temperatures from 16.0°C to 26.0ºC, representing temperatures that stress rearing fish. A score of 0 is given to temperatures above 26.0ºC, representing temperatures that reduce growth or kill fish. Temperatures below 12ºC are assigned scores from 4 to 0 in accordance with the function relationship developed by McMahon (1983) for overwintering smolts.

C.1.3.1.3 Steelhead Temperature Requirements

Delays in steelhead migration occur when natal streams are too warm (Major and Mighell 1966). Bjornn and Reiser (1991) report that upstream migrants prefer temperatures in the range of 10°C to 13ºC, while a CDFG report (1991) on Mokelumne River steelhead gives a preferred range of 7.8°C to 11.0ºC. Upstream migrating can stop altogether when temperatures reach 21.1ºC (Beschta et al. 1987, Cramer 1992) or falls below 4.0ºC. Based on these literature values, a score of 5 is given to temperatures of 7.8°C to 11.0ºC. A score of 0 is given to temperatures above 21.1ºC and below 4.0ºC, as these represent temperatures at which migration potentially ceases. The remaining temperature range is scored from 4 to 1 by interpolation between preferred and migration-inhibiting temperature values.

Successful spawning of steelhead is affected by water temperature before, during, and after spawning. Bell (1986) reports the preferred temperature range for Columbia River steelhead during spawning is 3.9ºC to 9.4ºC. A CDFG report (1991) for the American River reports a somewhat higher preferred range of 7.8ºC to 11.1ºC, which includes the effects of temperature on embryo survival. Raleigh et al. (1984) reports that brood steelhead should remain in water temperatures below 14.0ºC for 2 to 6 months before spawning, to ensure the production of good quality eggs. He also found that temperatures above 20.0ºC are unsuitable for spawning success. Based on the American River findings, a score of 5 is given to temperatures between 7.8ºC to 11.1ºC. A score of 4 is given to temperatures between 11.1ºC and 14.0ºC, as this is the temperature at which embryo quality declines (Raleigh et al. 1984). A score of 0 is given to temperatures above 20.0ºC and below 4.0ºC while the remaining scores are estimated by interpolation.

The preferred incubation temperatures for steelhead mirrors those for spawning, since the best scientific information available is the embryo survival data from the American River (CDFG 1991). Therefore, a score of 5 is given to temperatures between 7.8ºC to 11.1ºC. Since a temperature of 20.0ºC is unsuitable for spawning (Raleigh et al. 1984), it is given a score of 0. The lower temperature limit on embryo survival for steelhead is 1.5ºC (Raleigh et al. 1984), which is also given a score of 0. The remaining incubation scores are estimated by interpolation.

The preferred water temperatures for rearing juvenile steelhead on the American River are reported to range from 12.8°C to 15.6ºC (CDFG 1991), while Bell (1986) reports a somewhat lower preferred range of 10°C to 12.8ºC for steelhead in the Pacific Northwest. The experimentally established lower and upper ILT for steelhead are 0.0ºC and 23.9°C, respectively (Bell 1986, Bjornn and Reiser 1991). Hatchery-reared Central Valley steelhead consistently selected temperatures of 18 to 19°C, while wild fish, which were probably exposed to cooler temperatures in the Feather River, selected temperatures of about 17°C (Myrick and Cech 2000).

Temperature tolerances of steelhead in the southern portion of their range have not been well-documented and may be higher. In the Eel River, juvenile steelhead were observed feeding in surface waters with ambient temperatures up to 24.0°C (Nielsen et al. 1994). Roelofs et al. (1993) classified water temperatures in the Eel River as extremely stressful for steelhead above 26.0°C. Roelofs et al. (1993) report temperatures between 23.0°C and 26.0°C as causing chronic physiological stress that jeopardizes survival, and temperatures between 20.0°C and 23.0°C as producing chronic effects. A maximum weekly average temperature (MWAT) of 19ºC was calculated for steelhead by EPA (Brungs and Jones 1977, cited in Sullivan et al. 2000).

Steelhead use behavioral thermoregulation to survive stressful thermal conditions. For example, fish in streams and rivers utilize temperature gradients, such as thermal stratification in deep pools (Nielsen et al. 1994, Matthews et al. 1994).

Increases in water temperature will increase standard metabolism and food demand of salmonids. This demand can be met, up to a point, through higher water velocities, which can provide large amounts of drifting invertebrate food. Smith and Li (1982) reported that in Uvas Creek (Pajaro River watershed), steelhead eat more and maintain higher growths during high-flow regimes. By utilizing higher water velocity, and shallower and coarser substrate microhabitat, steelhead take advantage of portions of the water column substantially faster and more productively than at their resting positions. The fish make extensive use of cover to avoid higher water velocities and keep metabolic demand low, but use adjacent areas of sheer or "feeding lanes" to harvest drifting food.
Thus, steelhead eat more and maintain higher growth rates than they would in areas of slower velocity. Smith (1982) found that the density of trout in warmer stream reaches (19°C to 23ºC) was strongly dependent on water velocity, while in cooler stream reaches

(13°C to 17ºC), trout density was independent of water velocity. Warmer water temperatures will generally result in better growth at unlimited food up to a point, above which growth declines or fish lose weight.

Myrick and Cech (2000) investigated the effects of water temperature (10°C to 25ºC) on juvenile rainbow trout of the Eagle Lake subspecies and the Mt. Shasta strain to investigate the responses of different genetic strains to temperatures. No strain-related differences were found in conversion efficiency, oxygen consumption rates, thermal tolerance or swimming performance, but the Mt. Shasta strain trout grew faster at the highest temperatures (22 to 25ºC). Growth rates increased with temperature to a maximum near 19°C and declined at higher temperatures. Both strains were able to maintain weight at 25ºC for 30 days, which the authors suggest may allow them to survive short (<1 month) periods of sublethal temperatures in California streams.

Sullivan et al. (2000) completed a review of the effects of temperature on salmonids in the Pacific Northwest. They caution that careful consideration must be given to magnitude and duration of temperatures, and utilize a risk assessment approach to quantitatively estimate acute and chronic effect of temperature on salmonids. Their analysis suggested that there is little or no risk of mortality if annual 7-day maximum temperature is less than 26°C, but nevertheless suggest site-specific analyses be conducted when annual 7-day maximum temperature exceeds 24°C in local river conditions. Assuming an acceptable growth loss of 10 percent is an appropriate risk level, they suggest an upper threshold for the 7-day maximum temperature of 20.5°C is appropriate.

The NCRWQCB reviewed the water quality objective for temperature in the Russian River basin in Sonoma County to protect aquatic life, including listed species (NCRWQCB 2000). The review concludes that the upper lethal temperature for young steelhead is around 75º (23.9ºC), and that a maximum 7-day average stream temperature of 64ºF (17.8ºC) and a daily maximum temperature of 75ºF (23.9ºC) in the Russian River would likely protect the salmonid species present (including coho and Chinook salmonids). The report identified alternatives for the revision of the water quality objective for temperature, which are undergoing further review. The NCRWQCB report cautions that one of the difficult components to quantify is the effect of food availability on temperature tolerances of rearing salmonids, particularly for steelhead, as discussed in Smith and Li (1983). If primary and secondary production is high, then a numeric temperature objective specific to steelhead in the southern portion of their range may be higher than research based on colder climates would indicate. However, if food production is insufficient, higher temperatures could be detrimental (NCRWQCB 2000).

Myrick and Cech (2001) reviewed temperature effects on Central Valley steelhead. Located close to their southernmost range, Central Valley steelhead studies provide relevant information for the Russian River. Steelhead can be expected to show significant mortality at chronic temperatures exceeding 25ºC, although they tolerate temperatures as high as 29.6ºC for short periods of time. However, they experience sub-lethal effects at temperatures below these limits. Steelhead/rainbow trout acclimated to high temperatures tend to show greater heat tolerance than those acclimated to cooler temperatures (Cherry et al. 1977, Myrick 1998). Wild fish in thermal gradients selected temperatures around 17ºC, although the authors note that temperatures selected by Great Lakes rainbow trout increased with acclimation temperature from about 15ºC to 20º. Juvenile steelhead grow at temperatures ≤6.9°C to at least 22.5°C. The highest growth rates reported for Central Valley steelhead occurred at 19°C (Cech and Myrick 1999), but higher temperatures have not been tested. The ability of salmonids to tolerate elevated temperatures is a function of exposure time. The authors suggest that there may be physiological differences between California steelhead and those from more northern latitudes that result in different growth rates, but indicate that large-scale experiments are needed to draw clear conclusions.

As the Russian River watershed lies in the southern and warmer range of salmonid species, temperature criteria based on published values in colder climates would be conservative. Temperature criteria based on values from the American River are more likely to represent Russian River steelhead. A score of 5 is assigned to temperatures ranging from 12.8°C to 15.6ºC. A score of 0 is assigned to the experimentally derived lower and upper ILT values (Bell 1986, Bjornn and Reiser 1991), and remaining scores are estimated by interpolation. However, steelhead in the southern portion of their range may have higher temperature tolerances than studies in the Pacific Northwest suggest; therefore, these criteria may be conservative.

C.1.3.1.4 Chinook Salmon Temperature Requirements

McCullough (1999) found that migrating adult Chinook salmon die at temperatures of 21°C to 22ºC in the Columbia River. Observations in the San Joaquin River indicate that upstream migration is halted when temperatures exceed 21.1ºC (Hallock 1970, Cramer 1992). However, upstream migration of adult salmon has been noted in the lower Klamath at water temperatures as high as 24.4°C (Dunham 1968, cited in Boles 1988). Chinook salmon have been observed migrating upstream in the Russian River on days with a daily average temperature over 22.6°C (Chase et al. 2003). At the lower end of the temperature scale, Chinook salmon have been reported to migrate upstream in water temperatures as low as 3ºC (Bell 1986). Preferred upstream migration temperatures reported for summer-run Chinook salmon range from 13.9ºC to 20ºC, and for fall-run Chinook salmon range from 10.6ºC to 19.4ºC (Bell 1986). The Independent Scientific Group (1996), however, found that metabolic stress for migrating Chinook salmon increases when temperatures are greater than 15.6ºC, suggesting that preferred temperatures are lower than originally reported by Bell (1986). Based on these literature values, a score of 5 is assigned for water temperatures between 10.6°C to 15.6ºC. A score of 0 is assigned to temperatures greater than 21.1ºC, representing temperatures that published criteria suggest are lethal to migrating adults (Hallock 1970, Cramer 1992, Independent Science Group 1996). However, Chinook salmon in the southern portion of their range, including the Russian River (Chase et al 2003) may have higher temperature tolerances than studies conducted in the Pacific Northwest suggest. Criteria scores for temperatures between 15.6°C and 21.1ºC, and between 10.6°C and 0.8ºC, are estimated by interpolation.

While Chinook salmon can spawn in water temperatures ranging from 1ºC to 20ºC (Seymour 1956, Bell 1986), their preferred temperature range is much narrower (Bjornn and Reiser 1991). In British Columbia, Shepherd et al. (1986) observed that Chinook salmon adults typically spawn at temperatures between 10ºC to 17ºC. In the American River, Bell (1986) reports a lower preferred spawning range temperature of 5.6°C to 13.9ºC. Boles et al. (1988) found that embryo mortality of Sacramento River Chinook salmon increased dramatically when temperatures climbed above 16.7ºC and considers this temperature an upper limit for successful spawning. Based on findings in the American River, a score of 5 is assigned to temperatures between 5.6ºC to 13.9ºC. A score of 0 is assigned to temperatures above 16.7ºC (Boles et al. 1988) and below 1.0ºC (Seymour 1956, Bell 1986). The remaining scores were estimated by interpolation following the functional temperature relationships derived by McMahon (1983) for the Chinook salmon habitat suitability index model.

Optimal temperatures for incubation of Chinook salmon embryos depend on the water temperature adults are exposed to prior to spawning (Hinze 1959). Given these constraints, however, preferred incubation temperatures are thought to range from 4.4ºC to 13.3ºC (Bjornn and Reiser 1991) and 5.0ºC to 12.8ºC (Seymour 1956, Boles et al. 1988). Mortality rates for emerging fry increase by 20 percent when temperatures rise from 12.8ºC to 15.6ºC, and by at least 30 percent when temperatures reach 16ºC (Resource Agency Report 1989). Temperatures greater than 16.7ºC have been found to cause 100 percent mortality of Chinook salmon fry in the Sacramento River (Boles et al. 1988, Resource Agency Report 1989). A score of 5 was assigned to the preferred temperature reported by Seymour (1956) and Boles et al. (1988), as these are below the temperature values at which mortality rates of emerging fry increase. Scores of 4 and 3 are given for temperatures between 12.8ºC and 15.0ºC (Resource Agency Report 1989) and 2 and 1 for temperatures between 15.0ºC and 16.7ºC (Boles et al. 1988). Temperatures above 16.7ºC and below 1.0ºC are given scores of 0 (Boles et al. 1988, Resource Agency Report 1989, Raleigh et al. 1986). Criteria scores for temperatures between 5.0ºC and 1.0ºC are estimated by interpolation.

The upper and lower ILT during rearing of Chinook salmon juveniles have been determined experimentally. Brett (1952) estimated these values to be 0.8ºC and 26.2ºC, respectively, with spring-run Chinook salmon. He also reports a preferred rearing temperature range of 12ºC to 14ºC. Myrick and Cech (2002b) found that American River fall-run Chinook salmon, when fed at satiation rations, food consumption and growth rates increased as temperature increased over an 11 to 19°C range, but with insufficient food, juvenile salmon were not capable of maintaining condition over that range. Juvenile Chinook salmon showed positive growth at temperatures as high as 25°C (Brett et al. 1982, cited in Myrick and Cech 2001). A score of 5 is given to the preferred rearing temperatures (Brett 1952), while a score of 0 is assigned to lower and upper ILT values. The remaining rearing scores are estimated by interpolation.

C.1.3.2 Dissolved Oxygen

DO requirements vary with species, age, temperature, water velocity, activity level, and concentration of substances in the water (McKee and Wolf 1963, cited in Raleigh et al. 1984). As temperatures increase, DO saturation levels in the water decrease while the oxygen needs of the fish increase. Optimal oxygen levels for rainbow trout (the nonanadromous form of steelhead) appear to be ≥7 milligrams per liter (mg/l) at <15°C and ≥9 mg/l at >15°C (Raleigh et al. 1984). Incipient lethal levels of DO for adult and juvenile rainbow trout are approximately 3 mg/l, depending primarily on temperature.

Reduced concentrations of DO can reduce the swimming performance of migrating adult salmonids. Maximum sustained swimming speeds of juvenile and adult coho salmon at temperatures between 10°C to 20°C were reduced when DO dropped below air-saturated levels (approximately 8 to 9 mg/l at 20°C), and performance declined sharply when DO fell to 6.5 to 7.0 mg/l at all temperatures (Davis et al. 1963).

For embryos, the amount of oxygen available is influenced by flow through redds. Embryos are most sensitive to hypoxial conditions during their early stages of development (Alderdice et al. 1958, cited in Bjornn and Reiser 1991). While embryos may survive when DO concentrations are below saturation (but above a critical level), their development is often abnormal. Newly hatched steelhead and Chinook salmon alevins (yolk sac fry that hatch from the eggs and live for a brief period within the interstitial spaces of the streambed gravels) were smaller and weaker when incubated as embryos at low and intermediate DO concentrations than at higher concentrations (Silver et al. 1963). Reduced DO lengthened the incubation period of coho salmon embryos and they hatched as smaller alevins (Shumway et al. 1964, cited in Bjornn and Reiser 1991). In field studies, survival of steelhead (Coble 1961) and coho salmon embryos (Phillips and Campbell 1961) was positively correlated with intragravel DO in redds. Phillips and Campbell (1961) concluded that intragravel DO must average 8 mg/l for embryos and alevins to survive well. Bjornn and Reiser (1991) recommend that concentrations should be at or near saturation and that temporary reductions should drop no lower than 5.0 mg/l. The USFWS (Raleigh et al. 1986) recommends that for Chinook salmon, the lower limit of DO for survival with short-term exposures is ≥ 2.5 mg/l at temperatures ≤ 7°C; with optimal levels of ≥ 8 mg/l at temperatures between ≥ 7°C and ≤ 10°C, and ≥ 12 mg/l at temperatures >10°C.

Growth rate and food conversion efficiency in coho salmon juveniles are limited by DO concentrations less than 5 mg/l (Bjornn and Reiser 1991). Davis (1975) reviewed information on incipient DO response thresholds and has developed oxygen criteria related to concentration, water temperature, and percent saturation. Davis concluded that salmonids would not be impaired at concentrations near 8 mg/l (76 to 93 percent saturation) and that initial symptoms of DO deprivation would occur at approximately 6 mg/l (57 to 72 percent) saturation (Bjornn and Reiser 1991) (Table C-4). Because rainbow trout fry occupy habitat contiguous with adults, their DO requirements are assumed to be the same as adults (Raleigh et al. 1984). Bustard (1983, cited in Raleigh et al. 1986), reported that Chinook salmon juveniles survived with DO ranging from 3 to 7 mg/l. The USFWS has concluded that Chinook salmon juveniles can survive short-term exposures to 3 mg/l at temperatures ≤ 5°C, but optimal levels are ≥ 9 mg/l at ≤ 10°C and 13 mg/l at > 10°C.

Table C-4 Response of Freshwater Salmonid Populations to Three Concentrations of DO (Bjornn and Reiser 1991, Modified from Davis 1975)

Response	DO (mg/l)	Percent Saturation at Temperature (°C)					
		0	5	10	15	20	25
Function without impairment	7.75	76	76	76	76	85	93
Initial distress symptoms	6.00	57	57	57	59	65	72
Most fish affected by lack of oxygen	4.25	38	38	38	42	46	51

Criteria for DO are given in Table C-5. They are based on the literature cited and on the habitat suitability index models developed by USFWS.

Table C-5 DO Evaluation Criteria by Species and Life-History Stage

Coho Salmon				
	Nov 1 to Jan 31	Dec 1 to Mar 31	All Year	Feb 1 to May 15
Habitat Score	DO (mg/l) Upmigration	DO (mg/l) Spawning/ Incubation	DO (mg/l) Rearing	DO (mg/l) Downmigration
5	6.5	11.0	8.0	8.0
4	6.0	9.5	6.5	6.0
3	5.5	8.0	6.0	5.5
2	5.2	7.5	5.2	5.2
1	4.8	4.5	4.5	4.6
0	< 4.8	< 4.5	3.0	3.0
Steelhead				
	Jan 1 to Mar 31	Jan 1 to May 31	All Year	Mar 1 to Jun 30
Habitat Score	DO (mg/l) Upmigration	DO (mg/l) Spawning/ Incubation	DO (mg/l) Rearing	DO (mg/l) Downmigration (Juveniles)
5	6.5	9.0	8.0	8.0
4	6.0	7.3	6.5	6.0
3	5.5	6.5	6.0	5.5
2	5.2	5.9	5.2	5.2
1	4.8	5.4	4.5	4.6
0	< 4.8	< 5.0	3.0	3.0

Table C-5 DO Evaluation Criteria by Species and Life-History Stage (Continued)

Chinook Salmon

Habitat Score	Aug 15 to Jan 15 DO (mg/l) Upmigration	Nov 1 to Mar 31 DO (mg/l) Spawning/ Incubation	Feb 1 to Jun 30 DO (mg/l) Rearing	Feb 1 to Jun 30 DO (mg/l) Downmigration
5	6.5	11	8.0	8.0
4	6.0	9.5	6.5	6.0
3	5.5	8	6.0	5.5
2	5.2	7.5	5.2	5.2
1	4.8	4.5	4.5	4.6
0	< 4.8	< 4.5	3.0	3.0

C.1.3.3 TURBIDITY

An increase in sediment input and turbidity in a stream can create physiological or behavioral responses in salmonids. Sediment input can affect the survival of eggs. Turbidity reduces the amount of light that can penetrate water. Turbidity may affect salmonids indirectly by reducing primary productivity in a stream, which, in turn, affects the food web.

Although the terms “turbidity” and “suspended solids” are sometimes used interchangeably, the degree of turbidity does not always indicate the amount of particulate matter in the water. Turbidity is measured by the amount of light that penetrates the water and is measured in nephelometric turbidity units (NTUs) or Jackson turbidity units (JTUs). It is not a measure of the quantity or type of suspended matter, and similar concentrations of different types of suspended matter could result in different turbidity readings. An increase in suspended solids can be caused by increased sediment loading and sediment load can be measured in mg/l. Suspended sediment concentrations are more difficult to measure than turbidity. Equations have been developed to estimate suspended sediment concentrations from turbidity measurements, but researchers caution that relationships differ between drainages due to specific sediment characteristics (Lloyd et al. 1987).

In most streams, water is naturally turbid at times, usually when storms produce runoff. In fact, moderate levels of turbidity may give juveniles protection from predators. Turbidity levels of approximately 23 NTU apparently reduced the perceived risk of predation on juvenile Chinook salmon (Gregory 1993). Chinook salmon occupy turbid rivers for a significant portion of their early life.

High suspended-solid concentrations cause physiological and behavioral stress responses (Newcombe and MacDonald 1991), but lower concentrations may reduce predation on juveniles. Low or moderate exposures of short duration can be tolerated by the fish. In general, however, salmonids survive better in clear water at all lifestages, and high, long-term levels of turbidity can negatively affect them (Newcombe and Jensen 1996).

Newcombe and Jensen (1996) analyzed data from 80 studies on fish responses to suspended sediment in streams and estuaries. Categories of effects ranged from "no effect" to "behavioral and sublethal effects" to "lethal effects." Turbidity effects can range from behavioral effects like alarm reactions or avoidance responses to lethal and paralethal effects like reduced growth, delayed hatching, and mortality. In between these extremes are sublethal effects such as reduction in feeding, physiological stress, and poor condition.

Migrating salmonids avoid water with very high silt loads and will cease migration (Cordone and Kelley 1961, cited in Bjornn and Reiser 1991). However, they can migrate with high turbidity levels often associated with rainfall events.

Suspended sediment effects on eggs are not as clearly defined in terms of suspended solids as much as by the percent of fines in the gravels. When an excess of silt is deposited after spawning, intergravel flow is reduced and eggs can be "smothered". This results in a loss of DO, accumulation of catabolic waste products, and the promotion of fungal growth. Generally, 85 percent of salmon eggs will die if 15 to 20 percent of the voids are filled with sediment (Bell 1991).

Newly emerged fry are more susceptible to moderate turbidities than older fish (Bjornn and Reiser 1991). Turbidities in the 25 to 50 NTU range (equivalent to 125 to 275 mg/l of bentonite clay) reduced growth and caused more young coho salmon and steelhead to emigrate from laboratory streams than did clear water (Sigler et al. 1984). Larger juveniles and adults do not appear to be affected by ephemerally high concentrations of suspended sediments like those that occur during storms. However, juvenile coho salmon avoided water with turbidities exceeding 70 NTU (Bisson and Bilby 1982). Feeding and territorial behavior of juvenile coho salmon were disrupted by short-term exposures (2.5 days to 4.5 days) to turbid water (up to 60 NTU) (Berg and Northcote 1985). Juvenile salmonids tend to avoid chronically turbid streams (Lloyd et al. 1987) except when they use them as migration routes. Young salmonids subjected to continuous clay turbidities had lower growth rates than those living in clear water (Sigler et al. 1984).

Chronic turbidity decreases light penetration in streams, which can reduce primary productivity. Dramatic changes in light penetration and primary production can be caused by even small (5 to 10 NTUs) increases in turbidity above naturally clear conditions (Lloyd et al. 1987). By modeling the effect of various turbidity levels on light available at depth, Lloyd et al. (1987) calculated that a turbidity of only 5 NTUs can decrease the primary productivity of shallow, clear-water streams in Alaska by approximately 3 to 13 percent. An increase of 25 NTUs may decrease primary production by 13 to 50 percent. This can result in decreased production of zooplankton and macroinvertebrates (secondary production), and decreased abundance and production of fish (Lloyd et al. 1987). Lloyd, therefore, suggests a moderate level of protection for salmonids would be 25 NTUs above natural conditions in streams. A higher level of protection would be 5 NTUs above natural conditions, which would bring total turbidities in salmonid streams to 8 NTUs. Absolute turbidities of 8 NTUs and higher have been shown to reduce sport fishing in Alaska.

The Water Quality Control Plan for the North Coast Region sets a standard for turbidity as

> *"Turbidity shall not be increased more than 20 percent above natural occurring background levels. Allowable zones of dilution within which higher percentages can be tolerated may be defined for specific discharges upon the issuance of discharge permits or waiver thereof."*

The standard for suspended material is

> *"Waters shall not contain suspended material in concentrations that causes nuisance or adversely affect beneficial uses."*

It is difficult to determine what naturally occurring background levels are. The sediment criteria are not specific enough to protect salmonid habitats from the cumulative effect of sediment-related effects. Therefore, the NCRWQCB may propose numeric instream targets to integrate cumulative effects over annual timeframes instead of indicators that measure instantaneous conditions. The targets would be expressed as 10-year rolling average values (NCRWQCB 2000). The criteria proposed here are designed to be more specific than current NCRWQCB standards.

Criteria are developed for turbidity measured in NTUs for rearing and spawning habitat (Table C-6). A conversion ratio for NTUs to mg/l suspended solids may give only a rough estimate of suspended-solid concentrations in this watershed. It should be kept in mind that short exposures (a few hours to a couple of days) would have less of an effect than long exposures (a week or more). However, even short exposures of high turbidity (greater than 70 NTUs) can have severe effects.

For migration corridors, a less stringent set of turbidity criteria was developed. Since turbidity and suspended solids are thought to protect juveniles from predators, criteria for downstream migration have relaxed turbidity standards, up to the point that physiological stress is not excessive.

Table C-6 Evaluation Criteria for Turbidity

Score	Rearing Turbidity (NTU)	Juvenile Migration Turbidity (NTU)
5	< 10	< 25
4	10 – 15	25 – 50
3	15 – 25	50 – 60
2	25 – 50[1]	60 – 65
1	50 – 70[2]	65 – 70
0	> 70[3]	> 70

References:
[1] Sigler et al. 1984
[2] Berg and Northcote 1985
[3] Bisson and Bilby 1982

C.1.4 Fish Passage Criteria

Evaluation criteria are presented for fish passage at fish passage structures (fish ladders or restoration actions that improve fish passage) and for fish passage past diversion facilities.

C.1.4.1 Fish Passage Structures

Fish passage structures are usually required when anthropogenic structures such as culverts or dams block spawning runs. Deleterious effects on salmonids can occur during construction, operation, and maintenance of fish passage structures. These structures also have the potential to increase predation on protected species by concentrating predators (including fish, birds, and humans) and prey. Evaluation criteria for predation are presented in the next section.

Effects on fish passage may not have been considered during culvert design or installation. High-velocity water flows can severely restrict the upstream passage of fish, while low flows decrease water depth, thereby preventing upstream and/or downstream migration. For fish to swim through a culvert, a flow depth no less than the body width of the maximum-sized fish is required (Bell 1990). The length and slope of the culvert, combined with site-specific water velocities, also affect the ability of fish to swim upstream. Erosion at the outflow of the culvert can create a drop that blocks access to its entrance. If a splash apron is constructed to reduce erosion, it may also become a barrier to passage.

Dams may restrict fish passage in several ways. One way is to block passage directly. Another is to change the channel in such a way as to restrict fish migration. For example, Mumford Dam caused downcutting below its base to the point where upstream passage for anadromous fish was difficult or impossible over most flow conditions.

C.1.4.1.1 Evaluation Criteria for Fish Passage Design

Successful fish passage depends on several factors, including the species type, the developmental stage, the fishway entrance design, the style of fish passage used, and the rate of flow in the fishway. To ensure successful fish passage, the fishway must be carefully engineered in terms of width and depth relationships to provide the required low velocity flows. The design must also make it easy for fish to find the entrance with minimal or no delay. Finally, there must be enough water flowing through the fishway so that fish can find the entrance to the structure (attraction flow) and pass upstream with minimal delays.

While there are several types of designs, most fish passages contain some common design features. These features are summarized below (Bell 1990) and serve as a guideline for assessing design and operational issues. It should be noted that some of these features may not apply to fish passage designed to mimic natural conditions within a creek.

- Resting area velocities: 0.1 foot per second in pools, or a tenth of the normal swimming speed;
- *(either)* Maximum drop: 12 inches between pools;
- *(or)* Average maximum velocities over weirs or through orifices: 8 feet per second (fps);
- Entrance velocities into passage: 4 to 8 fps;
- Travel time through fishway: 2.5 to 4 minutes per pool;
- Space for fish in pool: 0.2 cubic foot per pound of fish; and
- Entrance eddies: recommended that cross-velocity not exceed 2 fps at 0 fishway discharge.

Achieving favorable attraction conditions is a function of such factors as fish behavior, river size, fishway flow strength, location of fishway entrance, and flexibility of the spillway operation. As river flows change, the relative importance of each component changes.

Site-specific conditions, especially tailwater hydraulics and channel width, determine entrance flow requirements. Fish will migrate along the channel banks during high flows to take advantage of lower flow velocities. Migrating fish will search laterally in combination with short fallbacks when confronted by a barrier. Low-flow entrances can be located close to the base of dams and should also be located beneath the nappe of the spillway when it separates a substantial distance from the dam. The location of the entrance should be at the upstream-most point of fish passage, and the location must take into account the locations where fish hold before attempting to pass the barrier. The entrance flow should be high enough to compete with spillway flow for fish attraction.

The jet of water leaving the fishway entrance is an extension of the fishway into the tailwater and it guides fish to the entrance. The greater the momentum of the jet, the further it reaches into the tailwater and the more successfully it can guide fish to the entrance. When a low-flow entrance is aligned perpendicular to the channel alignment, parallel to the barrier, or oriented at a small angle, the entrance jet penetrates the tailwater to a greater extent than if aligned perpendicular to the flow in the tailrace. When a high-flow entrance is placed at a low angle (30-degree angle), the protrusion into the stream of an angled entrance provides an abutment and velocity shadow behind which fish move upstream, and then passage is blocked by the abutment and high water velocities upstream of the entrance.

Attracting migrants to fish passages is particularly important during adult spawning runs when river flows are usually high. Insufficient attraction flows could make it difficult for adult fish to find the entrance to the fishway, thereby creating migration delays. In general, an attraction flow through the fishway system of 10 percent or more of the total streamflow is sufficient to guide fish. However, this 10 percent criterion can be impractical at very high flows, and therefore other fishway design factors (discussed above) can be considered. Because most of the streamflow passes through the fishways evaluated for this Draft BA, except during very high flows (storm events), evaluation criteria are not needed. Evaluation criteria for adult fish passage are presented for fish ladder design and operation.

Adult spawning migrations are induced by freshets. The fish passage structure should operate effectively at a range of flows that is wide enough to prevent substantial migration delays. Several design flow criteria have been developed. Gebhards (1972, cited in Bates 2000) suggests migration delays should not exceed 6 consecutive days. Dryden (1975, cited in Bates 2000) recommends that a 7-day migration delay should not be exceeded more than once in 50 years and a 3-day migration delay should not be exceeded during the average annual flood. CDFG suggests that passage should be provided during at least 90 percent of the flows that will be encountered for the target species during its migration period (Bates 1988).

Table C-7 lists scoring categories for the effectiveness of a fish passage design. High scores are given to fish passage facilities that meet the basic design criteria described above and pass fish with minimal or no delay.

Table C-7 Adult Fish Passage Evaluation Criteria Based on Fish Ladder Design and Operation

Category Score	Evaluation Categories
5	Fish passage passes adult salmonids without delay.
4	Fish passage passes adult salmonids with acceptable delay.
3	Fish passage passes all target species after extended delay.
2	Fish passage does not pass all target species of adult salmonids.
1	Passage provided but does not appear to pass any adult salmonids, or passage not provided.

Most published criteria address upstream spawning migrations, but juvenile downstream passage is also required. Juvenile downstream migrants have a finite amount of time to complete the physiological change (smoltification) that enables them to survive in a marine environment. A substantial delay in migration may result in smolts reverting to a resident form and they may spend an additional year in fresh water. Downstream migrant smolts need a minimum of 6 inches depth of water (Flosi et al. 1998). Furthermore, if a barrier decreases water velocity upstream, downstream passage through the fishway may be delayed or impeded entirely.

Restoration and conservation actions designed to restore fish passage may utilize design elements that improve both upstream and downstream passage for all life-history stages of salmonids, as well as for other species. They are likely to restore some of the natural functions inherent in an interconnected riverine ecosystem. Fish passage projects that restore the geomorphology of the stream are likely to have a greater benefit than a fishway that is designed primarily to pass adult salmonids. A fish passage project that removes a passage impediment or restores the geomorphology of a stream channel in a way that reestablishes good fish passage conditions would be beneficial to protected fish species.

C.1.4.2 FISH PASSAGE PAST DIVERSION FACILITIES

Death or injury of juvenile salmonids at water diversion intakes has been identified as a major source of fish mortality (National Marine Fisheries Service [NMFS] [now known as NOAA Fisheries] 1994). Improperly designed diversion facilities can cause impingement or entrapment of individuals and delay migration, and can result in stress-related injury or death.

When water in the infiltration ponds and flood control reservoirs becomes shallow and and/or stagnant, the likelihood of stress-related injuries to juveniles increases. There is also an increased potential for physical injury during fish rescue operations. In addition, if water levels drop too low, the flat bottom of a pond can cause stranding. Finally, if there is little cover in the ponds or reservoirs, predation rates of juveniles by predatory fish and avian species increase with decreasing water levels, and entrapped adults may be at a higher risk from poachers.

Evaluation criteria for fish passage at a diversion facility are based on 1) the proportion of surface water diverted, and/or 2) the degree of overlap between the migration period and the period of diversion operation.

In general, if more water is diverted during juvenile migration, the potential to affect fish increases. Chinook salmon and steelhead smolt tend to migrate downstream with rising flows, in addition to other factors (Mundy 1997). Their tendency is to move away from river margins (i.e., rearing habitat) and into mid- to near-surface water with increasing flow (McDonald 1960, cited in Northcote 1984), which may decrease their risk for entrainment or injury. It is estimated that if more than 50 percent of surface water flow is diverted, there is a significant risk of entrapment for salmonids. A score of 5 is given to facilities that do not affect surface flow during the migration period. A score of 1 is given to facilities that divert more than 75 percent of the surface water (Table C-8).

Table C-8 Juvenile Passage Evaluation Criteria — Opportunity for Entrapment, Impingement, or Injury During Operation — Amount of Water Diverted

Category Score	Evaluation Category
5	Facility does not affect any surface flow.
4	Facility diverts less than 25% of surface water flow.
3	Facility diverts between 25% to 50% of surface water flow.
2	Facility diverts between 50% to 75% of surface water flow.
1	Facility diverts more than 75% of surface water flow.

The effect of diversion timing on juvenile salmonid migration is evaluated by assessing the degree of overlap between the migration period and project operations. A score of 5 is given to facilities that do not affect surface water flow during any of the migration period, while a score of 1 is given to facilities that affect surface flows more than 25 percent of the time (Table C-9).

A diversion structure is often equipped with fish screens to prevent entrainment of young salmonids in diverted water. The effectiveness of these screens is evaluated to determine if salmonid juveniles or fry can pass the diversion without injury or delay.

Table C-9 Juvenile Passage Evaluation Criteria — Opportunity for Entrapment, Impingement, or Injury — Time Water is Diverted

Category Score	Evaluation Category
5	Facility does not affect surface water flow during any time of migration period.
4	Facility diverts surface water flow during less than 10% of migration period.
3	Facility operates between 10% and 15% of migration period.
2	Facility operates between 15% and 25% of migration period.
1	Facility operates during more than 25% of the migration period.

NOAA Fisheries has developed fish screening criteria for anadromous salmonids, for both fingerling and fry life-history stages (NMFS 1997). Fish screens at diversion intakes should meet NOAA Fisheries criteria at all flow conditions. NOAA Fisheries defines fry as less than 60 millimeters (mm) in length. The following is a summary of the more important fish screen elements recommended by NOAA Fisheries for salmonid fry and juveniles. The criteria are presented for 1) pump intake design, 2) river and canal screen design, and 3) escape or return mechanism design.

1. Pump intake design.

    - Approach velocity not to exceed 0.33 fps for fry and 0.8 fps for juveniles.
    - Sweeping velocity to be greater than approach velocity.
    - Perforated plate screen opening not to exceed 3/32 inch in diameter for fry and ¼ inch for juveniles, and have a minimum open area of 27 percent for fry and 40 percent for juveniles.
    - Face of screen surfaces to allow fish unimpeded movement parallel to screen face and ready access to bypass routes.
    - Structural features to protect fish screens from large debris.
    - Design features to eliminate (as possible) undesirable hydraulic effects (e.g., eddies, stagnant flow zones) that may delay or injure fish, or provide predator opportunities.
    - Screen and bypass to work in tandem to move outmigrating salmonids (including adults) to the bypass outfall with minimum injury or delay.
    - Bypass entrance to be provided with independent flow control.
    - Bypass entrance must equal or exceed the maximum velocity vector resultant along screen, upstream of the entrance.
    - Bypass entrance to extend from floor to water surface; require smooth interior pipe surfaces and joints; fish to not free-fall; pressure in the bypass to be equal or be above atmospheric pressure; fish to not be pumped within the bypass system.
    - Bypass system to minimize debris clogging and be accessible for cleaning; depth of flow 0.75 foot or greater; ambient river velocities at bypass outfall greater than 4.0 fps.
    - Bypass outfall to be located to minimize avian and aquatic predation; bypass located where there is sufficient depth to avoid fish injury; impact velocity not to exceed 25.0 fps, bypass outfall discharges to be designed to avoid adult jumping injuries.
    - Fish screens to be automatically cleaned; fish screen system to be evaluated for biological effectiveness and available for inspection to NOAA Fisheries.

2. River and canal screens.

    - Where practical, construct screen at diversion entrance.

- Make screen face generally parallel to river flow and aligned with adjacent bankline.
- Minimize eddies and undesirable flow patterns in the vicinity of the screen.
- Provide sufficient hydraulic gradient to route fish between trash rack and screen to safety.
- Provide screens downstream of diversion entrance with an effective juvenile bypass system to collect juvenile fish and safely transport them back to the river with minimum delay. The angle of the screen to flow should be adequate to effectively guide fish to the bypass.
- If fish are entrained within a canal or infiltration pond, escape or return to the river can mitigate some of the effects. Alternatives are provided as criteria below, in order of preference.

3. Fish escape or return mechanisms.
    - Provide a structure that returns the fish safely to the river prior to entrapment in a canal or pond.
    - Provide a structure that allows the fish to voluntarily return to the river after entrainment.
    - Provide rescue of entrapped fish which minimizes stress, injury, and death through rapid response (rescue within one week), and design and/or methods of capture and release that reduce potential physical injury.

Fish screens are evaluated according to their performance standards and ability to pass juvenile and fry-sized salmonids within NOAA Fisheries criteria (Table C-10). A score of 5 is given to facilities that meet NOAA Fisheries criteria and pass fish without delay. A score of 3 is assigned to facilities with fish screens that pose a moderate risk of entrapment, but have rescue or escape mechanisms to reduce mortality. A score of 1 is assigned to facilities without fish screens and rescue or escape mechanisms.

Table C-10 Juvenile Passage Evaluation Criteria for Screen Design

Category Score	Evaluation Category
5	Fish screens meet NOAA Fisheries criteria and pass fish without injury or delay.
4	Facility provided with fish screens, but the facility has a low risk of entrainment, impingement, or migration delay.
3	Facility provided with fish screens, but the facility has a moderate risk of entrainment, impingement, or migration delay; effective rescue or escape is provided.
2	Facility provided with fish screens, but the facility has a high risk of entrainment, impingement, or migration delay; ineffective rescue or escape is provided.
1	Facility not provided with fish screens; no rescue or escape is provided.

C.1.5 PREDATION CRITERIA

Structures that concentrate predators and prey, or introduce predators into salmonid habitat they have not previously had access to, have the potential to increase predation on protected species. Examples of such a structure include a fish passage structure or a dam.

Predators of particular concern are non-native largemouth bass and smallmouth bass, green sunfish, and native Sacramento pikeminnow. There are currently self-sustaining populations of these warmwater species in the Russian River. In stream areas that are easily accessible to people, structures may provide increased poaching opportunities.

C.1.5.1 EVALUATION CRITERIA FOR PREDATION

Structures that concentrate prey increase the potential for predation on protected species. If there are holding areas that favor predators near structures that concentrate salmonids, and if predators are actually present near those structures, protected species may be negatively affected. Structures that provide predators access to areas that they have not historically reached would increase the level of predation by introducing a new risk. However, structures that provide predators access to areas with established populations of predators may or may not increase the level of predation. Furthermore, water temperatures favorable to predators would be needed.

To evaluate the risk of increased predation on protected species, three components were developed for predation evaluation criteria: structural criteria, access criteria, and habitat criteria. Structural criteria (Table C-11) assess whether the structure concentrates predators and prey. Access criteria (Table C-12) assess passage opportunities for predators and whether predators are given access to areas they have not historically had access to. Predator habitat criteria (Table C-13) are based on water temperatures favorable to warmwater predators, especially centrarchids and Sacramento pikeminnow. The optimum temperature for Sacramento pikeminnow is 26.3°C (Knight 1985). Warmwater temperatures favor these predatory fish at the same time that they negatively affect protected salmonids and their ability to avoid predation.

Table C-11 Predation Evaluation Criteria: Structural (Component 1)

Category Score	Evaluation Criteria
5	No features that concentrate salmonids or provide cover for predators, concentrations of predators not found.
4	No features that concentrate salmonids, predator cover near, predators in low abundance locally.
3	Features that concentrate salmonids, no predator cover nearby, predators in medium to low abundance locally.
2	Features that concentrate salmonids, predator cover nearby, predators in medium to low abundance locally.
1	Features that highly concentrate salmonids, predators abundant locally.

Table C-12 Predation Evaluation Criteria: Access (Component 2)

Category Score	Evaluation Criteria
5	Structure does not allow passage of predators; predators not present near structure.
4	Structure does not allow passage of predators; predators present near structure.
3	Structure provides limited passage of predators, or limited passage to areas they are already well established, predators not present near structure.
2	Structure provides limited passage of predators to areas they have historically not been found or have been found in limited numbers, predators present in limited numbers near structure.
1	Structure provides passage of predators to areas they have historically not been found or found in limited numbers, predators present or migrate to structure.

Table C-13 Predation Evaluation Criteria: Warmwater Species Temperature (Component 3)

Category Score	Evaluation Criteria
5	Water temperatures < 13^{o}C
4	Water temperatures 13^{o}C – 18^{o}C
3	Water temperatures 18^{o}C – 20^{o}C
2	Water temperatures 20^{o}C – 22^{o}C
1	Water temperatures 22^{o}C – 24^{o}C
0	Water temperatures $\geq$ 24^{o}C

C.1.6 FLOOD CONTROL-RELATED CRITERIA

The USACE determines releases from Warm Springs Dam and Coyote Valley Dam from the flood control pool of the reservoirs. The magnitude and frequency of flow releases during flood control operations affect channel geomorphology, scour of spawning gravels, and the extent of bank erosion. The potential for these flood control activities to affect coho salmon, steelhead, and Chinook salmon habitat are discussed, and evaluation criteria are defined.

High flows are periodically required for channel maintenance to maintain variation in stream morphology important to habitat quality, such as meanders, pools, and riffles. Channel maintenance flows also serve to refresh spawning gravels by mobilizing the streambed and winnowing the fine sediments from the gravels. However, flood releases may affect spawning habitat by scouring gravels to a depth that destroys the egg pocket. Ideally, there would be a balance between periodic channel maintenance flows and stability of spawning gravels.

Flood control operations may affect channel geomorphology, including streambed and streambank stability and maintenance of channel equilibrium conditions (i.e., channel is neither aggrading or degrading over the long-term). Alterations in channel morphology can have a negative influence on fish habitat conditions. Bank erosion can increase sediment input that impairs spawning gravels by reducing the flow of oxygenated water and removal of metabolic wastes from redds. Alevins may also become entombed by fine sediment intrusion into spawning gravels. Pool habitat can be diminished by sedimentation. Streambank instability can reduce riparian vegetation. This results in a loss of cover habitat, increases in thermal loading by removing shade, and a reduction in the food supply by reducing the amount of terrestrial input. High-magnitude flood releases can scour spawning gravels, resulting in direct mortality of incubating embryos. Insufficient flows of moderate magnitude can alter the long-term balance of sediment supply and sediment transport, resulting in conditions of disequilibrium and channel aggradation.

Flood control releases from Warm Springs Dam and Coyote Valley Dam have reduced the magnitude of flood peaks in the Russian River drainage. Flood peaks are reduced when stored flood water is released over a longer period of time than would have naturally occurred. However, flood releases may still be of sufficient magnitude and duration to negatively affect spawning habitat if they scour gravels to a depth that destroys the egg pocket.

Sustained releases of flood flows have been cited as a potential cause of streambank instability on both Dry Creek and the mainstem Russian River. Prolonged discharges in excess of 2,500 cfs are believed to be responsible for accelerated bank erosion on Dry Creek (USACE 1999). Sustained releases of flood flows from Coyote Valley Dam are also cited as a contributor to streambank erosion on the mainstem Russian River (USACE 1999). However, for the mainstem Russian River there is no information on flow threshold at which bank erosion begins to occur. A flow threshold at which bank erosion was assumed to occur was developed and is described in Bank Erosion Evaluation Criteria in Section C.1.6.2.3. There are also no reports that specify which mainstem stream reaches are subject to erosion, except that "high sustained releases erode the river bank for miles downstream" (USACE 1998a).

C.1.6.1 CHANNEL MAINTENANCE/GEOMORPHOLOGY

The change in hydrologic regime associated with flow regulation by dams will influence channel geomorphic response (Collier, Webb and Schmidt 1966). The type and magnitude of adjustments depends on initial channel conditions and the extent of changes in discharge and sediment supply (Reiser and Ramey 1985). The effect of dams on the morphology of a river tends to diminish downstream due to discharge and sediment contributions from tributaries. Although the rate of channel change in response to flow regulation by dams is highly variable, most channel adjustments likely take place within a few decades following dam construction (Mount 1995).

Flow regulation by dam closure has reduced the magnitude of peak flood discharges at both Lake Sonoma and Lake Mendocino. In response, river channels typically modify their cross-sections by channel narrowing due to sediment deposition and encroachment by riparian vegetation. When the bed material is a sand and gravel mixture as on Dry Creek and the mainstem Russian River, channel incision will often accompany channel narrowing if the flood peaks are of sufficient magnitude to mobilize most of the bed materials. Excessive channel incision often results in over-steepened streambanks and subsequent erosion. If flood peaks are sufficiently reduced under flow regulation, then the coarser bed material will not be entrained, and only finer material will be transported, leading to an overall coarsening of the channel bed. At this point the river channel is armored, preventing further channel bed adjustments, although the streambank may remain susceptible to erosion. However, if flood peaks are substantially reduced so that there is little or no transport of coarse sediments, the channel response is likely to be aggradation. Coarse sediment supplied by local tributary input will exceed competence and lead to channel aggradation.

If flow regulation sufficiently reduces peak flood events so that the sediment transport regime is altered and coarsening of the channel bed or aggradation results, then fish habitat conditions may be negatively affected. Spawning gravels may be subject to accelerated rates of fine sediment intrusion, decreasing reproductive success. Increased sediment deposition in riffles may reduce benthic macroinvertebrate production, decreasing the available food base. Rearing habitat may also be affected due to sediment deposition in pools.

Dams and reservoirs interrupt sediment transport, which may lead to channel geomorphic changes. If a significant portion of the total sediment load is removed when coarse sediments are deposited within a reservoir, replenishment of sediments downstream will be reduced until there are sufficient sources of sediment input from downstream tributaries. This can lead to excess stream power immediately downstream of a dam, because relatively clear water with little sediment in transport can perform more work scouring sediments from the streambed, banks, and floodplain. Thus, entrainment of fine sediments below the reservoir may continue. Without sediment replenishment and with excess stream power, only the coarsest material may be left behind, leading to armoring of the channel bed.

To maintain channel geomorphic conditions, adequate flows are periodically needed to mobilize the streambed and transport sediments. Such flows are necessary to flush fine sediments from the streambed to provide suitable spawning and rearing conditions for salmonids. For example, since dam closure on the Trinity River, export of water and increase in sediment yields from the watershed have buried spawning habitat (Mount 1995). However, if flood releases are of sufficient magnitude and frequency to regularly scour redds, spawning may be negatively affected. This has occurred on the Sacramento River, where release of high peak discharges from Shasta Dam has led to widespread channel scouring and incision, leaving little spawning habitat and armored channels (Mount 1995). Ideally, there is a balance, or dynamic equilibrium, between periodic mobilization of the streambed, transport of sediment, and sediment deposition and stability of spawning gravels. Lack of peak flows can reduce spawning success, as can an increase in the frequency and magnitude of peak flows.

C.1.6.1.1 Changes Caused by Land-Use

The alteration of the flow regime associated with dams is not the only cause of changes in channel morphology. Land-uses and development in the Russian River watershed, including gravel extraction, agricultural practices, and urbanization, also influence channel geomorphic conditions. Clearing riparian vegetation, building roads, grazing, and other development activities can alter the flood hydrograph, increase bank erosion, increase sediment input from upland areas, and otherwise negatively influence channel geomorphic and aquatic habitat conditions. Land-uses that significantly increase or decrease (as in the case of gravel mining) sediment supply will cause as pronounced alterations in channel geomorphology as flood regulation by dams. Distinguishing the effects of flood-control operations separately from these land-use effects on channel conditions can be problematic.

Significant channel geomorphic changes were apparently already underway on Dry Creek before construction of Warm Springs Dam. A study conducted by USACE concluded that gravel mining on Dry Creek and on the mainstem Russian River had caused approximately 10 feet of incision along the 14-mile channel length by the mid-1970s (USACE 1987). The channel incision on Dry Creek initiated lateral instability and subsequent bank erosion so that channel width had increased from approximately 90 feet to over 450 feet in some locations in the 1970s (USACE 1987). The 1987 study concluded that it was unlikely that further channel degradation would occur, but that continued lateral instability and erosion of the incised channel banks was likely.

On the mainstem Russian River between Healdsburg and Ukiah, gravel mining has also recently altered channel geomorphic conditions. The East Fork Russian River had experienced up to 16 feet of channel bed degradation by the mid-1980s and in the Alexander Valley (near Cloverdale), approximately 2 feet of bed degradation had occurred by 1990 (EIP 1993).

C.1.6.2 Evaluation Criteria for Effects on Scour of Spawning Gravel, Streambank Erosion, and Channel Geomorphology

Evaluation criteria and analytical methods are described in detail for each of the channel geomorphic issues; scour of spawning gravel, streambank erosion, and channel geomorphology. All of the analyses consider how flood control operations affect those geomorphic issues. An important component of these analyses is related to expected streamflow conditions under flood control operations at Warm Springs Dam and Coyote Valley Dam. Streamflow has an important influence on channel geomorphic conditions and therefore on fish habitat.

Representative streamflow conditions were determined by using models rather than using actual, historic, streamflow data. The models provide a tool for simulating operational characteristics of the reservoirs and resulting streamflow conditions. As such, the hydrologic model emulates but does not necessarily match historic streamflow conditions exactly. The hydrologic model also has the advantage of being flexible. Operational conditions at each dam can be modified in the models so that streamflow conditions may be adjusted and the resulting potential change on geomorphic conditions and fish habitat tested.

For this report, two streamflow models were combined and used in the analyses. The SCWA model provides average daily flow at various locations along the mainstem Russian River between Coyote Valley Dam and Guerneville, and on Dry Creek downstream of Warm Springs Dam. The SCWA model was developed in the late 1980s to quantify relationships between streamflow, water demand, instream flow requirements, and water supply needs. The USACE HEC-5 model was developed specifically for this BA. This model also provides average daily flow at various locations along the mainstem Russian River between Coyote Valley Dam and Guerneville, and on Dry Creek downstream of Warm Springs Dam.

The results of the two models were combined so that flow conditions between June through October are derived from the SCWA model, and flow conditions between November through May are derived from the USACE model. The combined model results most accurately emulate historic flow conditions, since it was determined that the SCWA model did a better job of estimating low flow conditions, and the USACE model did a better job of estimating relatively high-flow conditions. The 36-year period of record covered by the combined model and used in all of the analyses are water years 1960 through 1995. For the remainder of this report, the combined flow model results are simply referred to as the hydrologic model.

C.1.6.2.1 Scour of Spawning Gravels Evaluation Criteria

Evaluation criteria for flood control effects on scour of spawning gravels were determined by estimating the hydraulic conditions necessary for the initiation of bed-particle motion. Incipient motion was derived from a modification of Shields' relationship for critical shear stress in non-uniform bed materials. Andrews (1983) determined that the average critical dimensionless shear stress for the median particle in the riverbed surface, $\tau *_{ci50}$, was 0.033. Andrews further found that in all rivers, the critical value of $\tau *_{ci50}$ was equaled or exceeded at the bankfull discharge. The mean bankfull dimensionless shear stress relative to the median particle diameter in the bed surface is 0.047. Thus, for an unarmored streambed, particles at least as large as the median diameter of the bed surface will be entrained by a bankfull discharge. Many hydraulic engineers and geomorphologists use 0.047 for critical dimensionless shear stress (Shields' parameter) in gravel bed streams (Simons, Li & Assoc. 1982), and this is the Shields value used in this assessment.

Using a Shields parameter of 0.047 for the mobilization of spawning-sized gravels on the bed surface, the Shields relationship for critical shear stress ($\tau *_{ci}$) is defined as:

$$\tau *_{ci} = 0.047 (\gamma_s - \gamma)d_{50}$$

Where:

γ, γ_s = specific weight of the fluid and sediment, respectively
$_{50}$ = median particle diameter

Thus, critical shear (the threshold at which incipient motion occurs) can be calculated for a particle size distribution with a known median diameter (d_{50}). There are no data available for the size of spawning gravels used by salmon and trout on either Dry Creek or the mainstem Russian River. For this analysis, the d_{50} (median particle diameter) of spawning-sized gravels was assumed to be as follows:

Coho Salmon	16 mm
Steelhead	22 mm
Chinook Salmon	36 mm

These d_{50} are based on a compilation of spawning gravel particle sizes reported from numerous studies on streams throughout the western states (Kondolf and Wolman 1993). The range of d_{50} represented by the 25th, 50th, and 75th percentile values from these studies are shown in Table C-14. The percentile values refer to the frequency distribution of d_{50} spawning gravel particle sizes associated with each species in the compiled studies. Thus, a 75th percentile value indicates that only 25 percent of the d_{50} particle size values exceed the value listed in Table C-14. The 50th percentile is synonymous with the median, indicating that one-half of the d_{50} particle sizes were greater than, and one-half less than, the listed values.

Table C-14 D_{50} Spawning Gravel Sizes Compiled by Kondolf and Wolman (1993)

	25th Percentile	50th Percentile	75th Percentile
Coho Salmon	12	16	30
Steelhead	18	22	32
Chinook Salmon	22	36	48

The critical shear stress calculated using a Shields parameter of 0.047 for the d_{50} of spawning gravels in coho salmon, steelhead, and Chinook salmon redds are shown in Table C-15.

Table C-15 Critical Shear Stress for Coho Salmon, Steelhead, and Chinook Salmon Spawning Gravels

	D_{50} (mm)	Shields Parameter	Critical Shear (lbs/ft^2)
Coho Salmon	22	.047	.349
Steelhead	16	.047	.254
Chinook Salmon	36	.047	.572

The critical shear stress was compared with values of actual shear stress for a range of flood-flow discharges on Dry Creek and the mainstem Russian River between Healdsburg and Ukiah in two distinct stream reaches: Alexander Valley and upstream of

Alexander Valley to Ukiah. Average shear stress values were determined for individual cross-sections using HEC-RAS hydraulic modeling. On Dry Creek, average bed-shear stress values were calculated for 112 cross-sections. These cross-sections were surveyed by SCWA. On the mainstem Russian River, 56 cross-sections located downstream of Coyote Valley Dam were used to determine shear stress values. Of the 56 cross-sections, 30 were surveyed in the Alexander Valley for the SCWA Aggregate Resources Mining Plan (SCWA 1999) in 1998 (Doris Anderson, SCWA, pers. comm.), and 26 were surveyed upstream of Alexander Valley to Ukiah in 1978 by Winzler and Kelly for USACE (1978). Since the effect of flood control operations from Coyote Valley Dam is insignificant below Healdsburg, and spawning is not considered to be significant on the lower mainstem reach (Winzler and Kelly 1978, Steiner 1996), no analysis was performed below Alexander Valley.

Actual shear stress values (calculated using HEC-RAS) that exceed the critical shear threshold identified in Table C-16 can be expected to initiate motion in redd gravels. Initiation of motion occurs when critical shear stress is exceeded by actual channel-bottom shear stress, although the transport rate and distance of transport of the streambed material may be quite small when the critical shear stress is only slightly exceeded.

To confirm that initiation of motion associated with a given discharge is likely to be sufficient to scour redd gravels to the depth of a typical egg pocket, a second, supporting analysis was performed to estimate depth of scour. The amount of scour in a riverbed depends on the ability of the bed to reform the surface layer after it has been ruptured. To make such a determination, the size of the streambed sediments must be known from field studies. The streambed particle size that will not move with a given discharge is determined. This is accomplished by comparing the critical shear stress needed to move a given particle size, (determined by the Shield's relationship), with the actual particle shear stress for that discharge. The actual particle shear stress can be derived from the velocity associated with the given flow. By observing the percentage of bed material less than the size of the maximum sediment which will not move, the depth of scour necessary to leave an armor layer can be calculated by the equation (Simons, Li & Associates 1987):

$$\Delta Z = 2\ d_a/1\text{-}P_c$$

where:

ΔZ is the depth of scour

d_a is the size of the armoring material

P_c is the percent of material finer than the maximum moveable size

The greater the percent of streambed material finer than the maximum moveable size, the greater the depth of scour. Conversely, the smaller the percent of streambed material finer than the maximum moveable particle size, the smaller the depth of scour.

The maximum moveable size of streambed material (d_a) was determined from a defined relationship between flow velocity and sediment size (EIP 1993, based Simons & Associates 1987). The average flow velocity at the discharge which initiates motion was determined from HEC-RAS model output for each cross-section. The average flow velocity is entered on the curve to determine the maximum moveable size of streambed material.

There are no available particle-size distribution curves from actually spawned gravels on either Dry Creek or the mainstem Russian River. For Dry Creek, the size distributions used to determine the percentage of bed material finer than the maximum moveable sediment size (P_c) is based on particle-size distribution data obtained from bed material grab samples (USACE 1987). Thirteen particle-size distribution curves, each representing a different location along the Dry Creek channel profile, were developed from USACE 1987 data and from twelve particle-size distribution curves developed from USACE 1999 data. For the mainstem Russian River, Alexander Valley to Ukiah, five particle-size distribution curves developed from recent 1999 bulk sampling in riffles performed by SCWA were used.

The streambed degradation analysis provides an estimate of depth to which scour will occur, confirming if redds are likely to be disturbed to the depth of the egg pocket. The average egg pocket depth for coho salmon, steelhead, and Chinook salmon is 20 to 30 centimeters (cm) (7.9 to 11.8 inches) (Bjornn and Reiser 1991). For this analysis, depth to the egg pocket was assumed to be 8 inches (0.7 feet) (B. Cox, CDFG, pers. comm. 2000).

The analysis for influence of flood operations on scour of spawning gravels is based on the following procedure:

1) Shear stress values determined in the HEC-RAS model are compared with the critical shear thresholds defined in Table C-15 and used to determine the discharge at which initiation of motion will occur. The number of cross-sections expected to have initiation of motion for specified flow ranges are identified in (Tables C-16 to C-22). The number of cross-sections at which spawning-sized gravels will likely not experience initiation of motion given the existing hydrologic regime is also identified in each table. Three tables are presented for Dry Creek, one for each species. On the mainstem Russian River only steelhead and Chinook salmon gravels are evaluated, since coho salmon do not spawn on the mainstem.

Initiation of motion associated with the range of discharges is plotted as cumulative curves for each species using Tables C-16 to C-22. The initiation of motion curves show the cumulative number of cross-sections at which shear stress exceeds critical shear.

2) The number of flood events that occur in the designated flow categories (Tables C-16 to C-22) are tallied for the period of record, water years 1960 to 1995, derived from flow modeling. The flow modeling represents the range of streamflow conditions expected under present-day flood control operations of Warm Springs Dam and Coyote Valley Dam. The flow modeling is not an evaluation of actual historic conditions, but rather a

tool which characterizes the magnitude and frequency of representative runoff conditions over time.

3) An ordinal ranking score is applied to all flood events for each of the defined flow categories based on the different time periods when the flows occur and based on each of the three fish species of concern. The ordinal ranking score, 1 to 5, assigns a 1 to the highest potential effect and a 5 to the lowest potential effect. High potential for effects (i.e., low ordinal ranking) was assigned to higher flows and flows which occur during the latter part of the spawning and incubation season. Those flows have the greatest potential to scour the most redds and incubating alevins. The criteria for scoring are defined for each of the stream reaches and for each species as shown in Tables C-23 to C-29.

4) Depth of scour is calculated to determine if the 0.7-foot criterion is exceeded for the discharge range associated with initiation of motion.

<u>Dry Creek</u>

Table C-16 Steelhead Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-Sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<1,300 cfs	25	22
>1,300-2,600 cfs	27	46
>2,600-5,500 cfs	32	75
>5,500 cfs	24	96

Never moved: 4 cross-sections = 4% of total 112

Table C-17 Chinook Salmon Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-Sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<3,000 cfs	21	19
>3,000-6,000 cfs	25	41
>6,000-9,000 cfs	20	61
>9,000 cfs	18	79

Never moved: 24 cross-sections = 21% of total 112.
Note: discharge greater than 8,000 cfs has not occurred on Dry Creek.

Table C-18 Coho Salmon Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-Sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<800 cfs	28	25
>800-1,400 cfs	27	49
>1,400-3,000 cfs	29	75
>3,000 cfs	26	98

Never moved: 2 = 2% of cross-sections.

Mainstem Russian River in Alexander Valley

Table C-19 Steelhead Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-Sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<2,000 cfs	7	23
>2,000-5,000 cfs	8	50
>5,000-12,000 cfs	7	73
>12,000-24,000 cfs	7	97

Never moved: 1 cross-section = 2% of total 30.

Table C-20 Chinook Salmon Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-Sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<5,000 cfs	6	20
>5,000-18,000 cfs	9	50
>18,000-27,000 cfs	7	75

Never moved: 8 cross-sections = 25% of total 30.

Mainstem Russian River Upstream of Alexander Valley to Ukiah

Table C-21 Steelhead Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<500 cfs	10	38
>500 cfs	7	65

Never moved: 9 cross-sections = 35% of total 26.

Table C-22 Chinook Salmon Spawning Gravels: Number of Cross-Sections with Initiation of Motion

Flow Range	Number of Cross-sections with Initiation of Motion in Given Flow Range	Cumulative Percent Moved (%)
<1,000 cfs	5	19
>1,000 cfs	4	38

Never moved: 16 cross-sections = 62% of total 26.

Scoring

The scoring system shown in Tables C-23 to C-29 is based on the number of cross-sections that will initiate bed movement within each of the stream reaches evaluated. As flows increase and more cross-sections experience bed movement, scores are lower. Whenever possible, at approximately every 20 percent to 25 percent incremental change in the number of cross-sections moved, the corresponding ordinal ranking scores are lowered by 1. Thus, the first 20 percent of the cross-sections moved in the given flow range is given a 5, the next 20 percent (i.e., cumulative of 40 percent moved) receives a 4, and so on. Scores do not go to 0 at any of the locations because there were always some cross-sections at which shear values never attain the critical shear threshold, so there is no initiation of motion. This occurred at several of the most upstream cross-sections on the mainstem Russian River where large streamflows overbank and fill the floodplain before critical shear is attained.[1] This also occurs at some of the wider cross-sections that do not obtain sufficient depth of flow to generate the shear stress necessary to initiate motion of spawning-sized gravels. Ordinal ranking scores do not reach the lower values when a relatively large percentage of the cross-section's shear values do not exceed critical shear threshold over the flow range (e.g., see Tables C-28 and C-29).

The first time-period in each of the tables below is the estimated period before spawning is over, and the second estimated time-period is during incubation after spawning is over.

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1 This is one important function of floodplains. By allowing overbank flows, there is "hydraulic release," limiting the magnitude of bed shear stress.

Scores are lower during the incubation time-period to reflect the fact that flows which disrupt spawning gravels with incubating eggs will likely have a greater negative effect on reproductive success for that year's class. Each of the daily flows from the hydrologic modeling record was scored for the relevant spawning and incubation time periods. The final score given for each water year is the highest impact event that occurs during the year.

Dry Creek

Table C-23 Coho Salmon Scoring Criteria for Scour of Redds in Dry Creek

Flow Range	Coho Salmon Dec 1–Jan 31 (before spawning is over)	Coho Salmon Feb 1–Feb 28 (incubation)
<800 cfs	5	5
>800 – 1,400 cfs	4	3
>1,400 – 3,000 cfs	3	2
>3,000 – 8,700 cfs	2	1

Table C-24 Chinook Salmon Scoring Criteria for Scour of Redds in Dry Creek

Flow Range	Chinook Salmon Nov 1-Jan 31 (before spawning is over)	Chinook Salmon Feb 1-Mar 31 (incubation)
<3,000 cfs	5	5
>3,000 – 6,000 cfs	4	3
>6,000 – 9,000 cfs	3	2
>9,000 – 15,000 cfs	2	1

Table C-25 Steelhead Scoring Criteria for Scour of Redds in Dry Creek

Flow Range	Steelhead Dec 1-April 30 (before spawning is over)	Steelhead May 1-May 31 (incubation)
<1,300 cfs	5	5
>1,300 – 2,600 cfs	4	3
>2,600 – 5,500 cfs	3	2
>5,500 – 12,000 cfs	2	1

Mainstem Russian River in Alexander Valley

Because coho salmon do not utilize the mainstem Russian River for spawning, only scour of Chinook salmon and steelhead spawning gravels were evaluated.

Table C-26 Chinook Salmon Scoring Criteria for Scour of Redds in Alexander Valley

Flow Range	Chinook Salmon Nov 1-Jan 31 (before spawning is over)	Chinook Salmon Feb 1-Mar 31 (incubation)
<5,000 cfs	5	5
>5,000 – 18,000 cfs	4	3
>18,000 – 27,000 cfs	3	2

Table C-27 Steelhead Scoring Criteria for Scour of Redds in Alexander Valley

Flow Range	Steelhead Dec 1-April 30 (before spawning is over)	Steelhead May 1-May 31 (incubation)
<2,000 cfs	5	5
>2,000 – 5,000 cfs	4	3
>5,000 – 12,000 cfs	3	2
>12,000 – 24,000 cfs	2	1

Mainstem Russian River Upstream of Alexander Valley to Ukiah

Table C-28 Chinook Salmon Scoring Criteria for Scour of Redds in the Upper Mainstem Russian River

Flow Range	Chinook Salmon Nov 1-Jan 30 (before spawning is over)	Chinook Salmon Feb 1-Mar 30 (incubation)
<1,000 cfs	5	5
>1,000 cfs	4	3

Table C-29 Steelhead Scoring Criteria for Scour of Redds in the Upper Mainstem Russian River

Flow Range	Steelhead Dec 1-April 30 (before spawning is over)	Steelhead May 1-May 30 (incubation)
<500 cfs	5	5
>500 cfs	4	3

Because the effect of flood control operations from Coyote Valley Dam is insignificant below Healdsburg, and spawning is not considered to be significant on the lower mainstem reach (Winzler and Kelly 1978, Steiner 1996), no analysis was performed below Alexander Valley.

C.1.6.2.2 Bank Erosion Evaluation Criteria

On Dry Creek, criteria for evaluation of streambank stability effects are based on an analysis of the frequency of flood flows greater than 2,500 cfs. Prolonged discharges in excess of 2,500 cfs are responsible for accelerating bank erosion on Dry Creek (USACE draft Biological Assessment 1999). Daily average flow from the hydrologic model was used for the assessment. For each year in the period of record (1960-1995), flows greater than 2,500 cfs were tallied. Scoring is based on the percentage of time in each water year that exceeds 2,500 cfs, as shown in Table C-30. The greater the number of days in any given year with flows exceeding 2,500 cfs, the lower the score. For years with flows greater than 2,500 cfs occurring less than 1 percent of the time (i.e., 3 days or less per year), a score of 5 is applied. For years with 2,500 cfs or greater magnitude flows occurring more than 4 percent of the time in any given year (16 or more days), a score of 1 is applied.

Table C-30 Evaluation Criteria for Dry Creek Streambank Stability

Percent of Time Flows Greater than 2,500 cfs	Number of Days per Year	Score
<1%	3 or less	5
1% – 2%	4 – 7	4
>2% – 3%	8 – 11	3
>3% – 4%	12 – 15	2
>4%	16 or more	1

No flow threshold has been specified at which bank erosion occurs on the mainstem Russian River. Therefore, the same unregulated recurrence interval flood that initiates bank erosion on Dry Creek was selected as the flow at which bank erosion is initiated on the mainstem below Coyote Valley Dam. On Dry Creek, the flow which initiates bank erosion, 2,500 cfs, corresponds to an 88 percent exceedance flow (as a 1-day annual maximum) or a 1.1-year, 1-day recurrence interval flood under unregulated conditions. This is slightly greater than the annual flood, which over the long-term will be equaled or exceeded approximately once every year. The 1.1-year, 1-day flood under unregulated conditions is 6,000 cfs at Hopland and 8,000 cfs at Cloverdale.

The analytical approach for flood operation effects on mainstem Russian River bank erosion is the same as for Dry Creek, using 6,000 cfs at Hopland and 8,000 cfs at Cloverdale. Scoring criteria for both locations are shown in Table C-31. Streambank erosion was not considered further downstream since the ability to control flood flows becomes greatly diminished at Healdsburg.

Table C-31 Scoring Criteria for Mainstem Russian River Streambank Stability

Percent of Time Flows >6,000 cfs at Hopland and >8,000 cfs at Cloverdale	Number of Days per Year	Score
<1%	3 or less	5
1% – 2%	4 – 7	4
>2% – 3%	8 – 11	3
>3% – 4%	12 – 15	2
>4%	16 or more	1

Flow changes above 1,000 cfs/hr are generally limited to a rate of 1,000 cfs/hr (interim ramping guidelines) to protect against bank sloughing and are not related to fish stranding issues. There may be a relationship between the rate at which flows are ramped down and the potential for saturated streambanks with high-pore pressures to slough. However, there are no data available on either Dry Creek or the mainstem Russian River to relate high flow recession rates to incidences of bank erosion.

C.1.6.2.3 Channel Maintenance/Geomorphology Evaluation Criteria

There is no single, well-established methodology to determine how regulated flood flows may change channel geomorphology or affect fish habitat. An equilibrium channel morphology (stream channel is neither aggrading nor degrading over the long-term) is maintained by flows that mobilize the streambed surface, transporting bedload at a rate is approximately equal to sediment supply. Maintaining the frequency of incipient motion of the channel bed is often used as a minimum criteria for maintenance of channel morphological conditions. It is characteristic for alluvial channels to have incipient mobilization of the channel bed at discharges that are approximately 80 percent of the 1.5- to 2.0-year annual maximum flood stage height (bankfull stage) (Andrews 1983). Typically, the 1.5- to 2.0-year annual maximum flood is considered to be the flow which, over the long-term, will do the most work in transporting sediments and is therefore defined as the effective or “channel-forming” discharge (Leopold 1994).

Maintenance of geomorphic conditions is based on the channel-forming 1.5-year annual maximum flood flow, shown in Table C-32. The 1.5-year flow can be expected to occur approximately twice out of every 3 years, or 66 percent of the time. Thus, for the 36-year period of record available from the hydrologic model, there should be approximately 24 flood flows that occur as annual peaks that equal or exceed the 1.5-year flood.

Scoring criteria consider how often the flow regime equals or exceeds the natural channel-forming discharge (1.5-year annual maximum flood flow). If the current flow regime achieves or exceeds the natural 1.5-year annual maximum flood magnitude in approximately two-thirds of the years over the simulated period of record (approximately 24 years out of 36 years), then channel maintenance is maximized, and the score is 5. If the current flow regime does not meet or exceed the natural channel-forming flow as frequently, then channel maintenance is not maximized, and lower corresponding scores are given.

The hydrologic modeling provides a simulated representation of average daily flow for the period of record. The 1.5-year channel-forming flow is calculated based on the annual instantaneous peak discharge, which will always be greater than the average daily flow. Therefore, to perform this assessment, it was necessary to estimate the average daily flow that corresponds to the 1.5-year instantaneous peak discharge. The corresponding average 1-day discharge was previously calculated by USACE (1998b), and is shown in Table C-32. The 1-day, 1.5-year annual flood flow is used as the criteria for this analysis. The assumption is that the 1-day flood flow includes the instantaneous peak flow that corresponds to the channel-forming discharge. This assumption may not be strictly true close to the dams, because flood-flow releases are controlled and relatively evenly distributed throughout the day (Paul Pugner, USACE, pers. comm. 2000). However, with distance downstream from the release point, the contributing drainage area will make up an increasingly larger proportion of the streamflow, resulting in higher instantaneous peaks contained within the average daily discharge.

Table C-32 Channel Maintenance Flow Associated with the 1.5-Year Peak Discharge and 1.5-Year 1-Day Discharge

	1.5-Year Peak Discharge	1.5-Year 1-Day Discharge
Dry Creek below Warm Springs Dam	9,500	5,000
Dry Creek near Geyserville	11,000	7,000
Russian River at Hopland	14,500	9,500
Russian River at Cloverdale	18,000	14,000
Russian River at Healdsburg	25,000	21,000

Note: 1.5-year unregulated flow for peak and 1-day discharge from USACE flood-frequency curves.

Scoring criteria are shown in Table C-33. A single score is given for the entire period of record (water years 1960 to 1995). Any single-year alone does not encompass a sufficiently long time-period to assess whether flood control operations are adequate to maintain channel geomorphic conditions. By definition, the channel-forming flow should occur approximately twice out of every 3 years, as a long-term average. When the channel-forming flow occurs less frequently, lower scores are applied. If the maximum annual discharge never meets or exceeds the threshold for the natural channel-forming flow, the score is 0. Channel-forming flows that occur more frequently received correspondingly higher scores (see Table C-33). The scoring applies equally to coho salmon, steelhead, and Chinook salmon.

Table C-33 Scoring Criteria for Maintenance of Channel Geomorphic Conditions

Proportion of Years with Channel Maintenance Flows	Number of Years per 36-Year Period of Record[a]	Score
51% – 66%	19 – 24	5
36% – 50%	14 – 18	4
21% – 35%	8 – 13	3
11% – 20%	5 – 7	2
1% – 10%	4 or less	1
0%	0	0

[a] Multiple channel-forming flows that may occur in a single year are counted as one occurrence for that year.

C.1.7 Fish Stranding Criteria

Ramping rates (reductions in flow) during dam maintenance activities or flood control operations have the potential to strand fish.

Recent research in Washington indicates that natural flow recessions associated with the annual snowmelt hydrograph occur at a very slow rate and tend to reduce the likelihood of stranding of small salmonids (Hunter 1992). If discharge is decreased too rapidly by flow regulation, then juvenile, or even adult salmon, can be stranded and killed. Project operations that have the potential to cause rapid flow fluctuations include operations at Coyote Valley Dam, Warm Springs Dam, and the inflatable dam at Mirabel.

Juveniles, particularly fry, are more susceptible to stranding than adults. Once Chinook salmon grow 50 mm to 60 mm or steelhead grow to 40 mm, they are substantially less vulnerable, but adult stranding has also been documented (Hunter 1992). Fry that have just absorbed the yolk sac and have recently emerged from the gravel are the most vulnerable because they are poor swimmers and typically reside along shallow stream margins (Phinney 1974, Woodin 1984). Stranding of juvenile coho salmon and rainbow trout on a gravel substrate in an artificial stream at low temperature was less frequent at slow rates of dewatering (6 cm/hr stage change rather than 30 cm/hr) and if flow reductions occurred at night (Bradford, et al. 1995). Stranding of juvenile coho salmon was reduced when the slope of the bar exceeded 6 percent.

The behavioral response of fish to flow fluctuations and how it may cause downstream emigration is not well understood. Studies conducted during the early 1970s by McPhee and Brusven (1976, cited in Hunter 1992) demonstrate that streamflow fluctuations trigger benthic drift and cause juvenile salmon to migrate downstream. Streamflow fluctuations can also cause both juvenile and adult fish to become trapped in shallow areas that are then exposed to elevated temperature or predation.

Redds are also susceptible to lowering water levels. Salmonid eggs can survive for weeks in dewatered gravel if they remain moist and are not frozen or subjected to high temperatures. However, dewatering is lethal to alevins. Since salmonids spawn over a period of months, eggs and alevins are often present at the same time.

Ramping rates typically constrain the rate (cfs/hr) at which a controlled release can be changed. Ramping rates are important to fisheries management agencies because they affect the rate at which instream hydraulic, and therefore habitat conditions, can be changed. The rate at which a controlled release is changed affects the rate at which total streamflow and downstream flow depths, flow velocities, channel top widths, and wetted surface areas change. The degree to which a particular ramping rate affects instream hydraulic and habitat conditions depends upon several site-specific factors:

- Percentage of total streamflow affected by the ramped release
- Amount of streamflow during ramping
- Stream channel shape, cross-sectional area, and slope
- Downstream distance from the ramping location

Perhaps the most difficult factor to understand quantitatively is the degree to which a flow change is "attenuated" as it progresses downstream. The influence of a sudden change in flow on stage is most pronounced at the location where the change occurs and decreases rapidly in the downstream direction. If a controlled release is ramped up, a portion of the released water goes into channel storage rather than directly into streamflow. Channel storage is represented by that portion of the channel cross-section over which the increased flow is spread, or temporarily "stored," along the channel length. This reduces the amount of flow and moderates the resulting change in water surface elevation (WSE) (stage) observed downstream from the point of ramping. If the controlled release is ramped down, a portion of channel storage is "evacuated" to become streamflow. The rate and degree to which channel storage changes influence stage primarily depends upon the size of the flow change (ramping) relative to streamflow and channel size, cross-sectional area, channel shape, and slope. Tributary inflow is also important. As tributary inflow contributes to streamflow in the channel, the relative effect of ramping represents a proportionally smaller influence on total channel flow and associated change in stage.

For analysis of ramping rates on Dry Creek, attenuation is assumed to occur within 1.0 to 1.5 miles downstream of Warm Springs Dam, which is the location of the first major tributary input at Pena Creek. On the mainstem Russian River, ramping effects are assumed to be attenuated by approximately 5 miles or less downstream of Coyote Dam near the Perkins Street bridge crossing in Ukiah. At the Forks, there is usually considerable flow from the mainstem Russian River during flood control operations that would attenuate ramping effects. Flows of approximately 2,500 cfs on the mainstem Russian River influence backwater effects on the East Fork (Pugner, USACE, pers. comm. 2000). Flow in the mainstem Russian River is usually increasing as reservoir releases are reduced during flood control operations, which moderates the ramping effects.

Table C-34 outlines the periods when salmonid fry may be present. Rearing coho salmon and steelhead fry may be present in Dry Creek in late winter and spring. Additionally, steelhead and coho salmon juveniles may be present in Dry Creek. In the mainstem Russian River below Coyote Valley Dam, Chinook salmon and steelhead fry as well as coho salmon, steelhead, and Chinook salmon juveniles, may be present during various times in the year. The critical issues addressed for operations of Coyote Valley Dam are reduced instream flow effects on habitat conditions and the potential for stranding below the dam. Below Warm Springs Dam, the critical issue is reduced streamflow effects on habitat conditions.

Table C-34 Times When Fry May Be Present in the Russian River Drainage

Species	Emergence	Fry may be present
Coho Salmon	Feb 1 – March 31	Feb – April
Steelhead	March 1 – May 31	March – June
Chinook Salmon	Feb 1 – March 31	Feb – April

Stress is likely to occur when fish are displaced from established rearing areas and crowded into residual pools. Residual pools with high fish densities could be subject to food competition, or predation by avian species and vertebrates, including hatchery fish preying on wild fish. Stranding could occur on riffles, gravel bars, and in backwater pools if flow becomes intermittent, and mortality may result if fish become desiccated. Water temperatures could also be elevated.

The Washington Department of Fisheries has proposed a rate of stage change that will generally protect fish (Hunter 1992). Hunter's ramping guidelines are modified with the phenology of salmonids in the Russian River Basin for this assessment (Table C-35).

Table C-35 Rates of Stage-Change Based on Hunter (1992) and Life-History Stages for Salmon and Steelhead in the Russian River Basin

Season	Rates
March 1 to July 1	1 inch/hour (0.08 foot/hour)
June 1 to November 1	2 inches/hour (0.16 foot/hour)

Drawing from Hunter's proposed guidelines, during juvenile rearing periods, which occur year-round for steelhead and coho salmon in the Russian River Basin, 2 inches/hour (0.16 foot/hr) stage change is appropriate. In the Mirabel Rubber Dam Fish Sampling Program (Chase et al. 2000), Data from SCWA's sampling program at Mirabel provide and indication of the size of salmonids in this portion of the mainstem (Chase et al. 2000, 2003). Chinook salmon averaged approximately 35 to 40 mm FL during the first few weeks of their life in 2002, then quickly grew to an average of approximately 80 mm by mid-April. The large numbers of steelhead YOY observed in 2002 (as in 2000) suggests that steelhead spawn and rear in the mainstem Russian River. Steelhead YOY became abundant in mid-April 2002 at an average FL of approximately 40 mm. The average size of steelhead YOY increased form 44 mm to 84 mm between April and June 2000. A few steelhead YOY captured in the Wohler Pool during August 2000 electrofishing surveys were generally larger than similar aged steelhead captured in Mark West and Santa Rosa creeks during fall surveys (Chase et al. 2000), suggesting mainstem-reared fish may have higher growth rates.

The Hunter (1992) guidelines are considered to represent a rigorous and conservative ramping standard for the Russian River watershed. Hunter developed his guidelines based on streams located in the northwest, a hydrologic regime that is dominated by snowmelt processes. Snowmelt streams usually have relatively gradual changes in runoff conditions. In the Russian River drainage, streamflow is driven by often intense Pacific frontal storms that naturally result in very “flashy” runoff conditions and therefore relatively larger changes in stage compared with snowmelt runoff conditions.

A comparison of the Hunter guidelines with natural flow recessions following storm events in the Russian River demonstrates this point. Stage changes associated with the receding limb of storm events were reviewed for the U.S. Geological Survey (USGS) Ukiah gage (11461000) located above the Forks and were compared to the Hunter guidelines (Figure C-1). For the period November 1995-June 1999, the average stage change is approximately 0.3 to 0.4 foot/hr when flows are greater than 1,500 cfs. At the 90^{th} percentile, stage changes range from 0.4 to 0.5 foot/hr or more when flows are greater than 1,500 cfs.

C.1.7.1 Flood Control Operations

To protect spawning gravel and juvenile salmonids within the Russian River and Dry Creek during flood control operations, USACE, in consultation with NOAA Fisheries and CDFG, developed interim guidelines for flow release changes. Proposed ramping rates for low reservoir outflows (0 to 250 cfs) are lower. These ramping rates summarized as follows:

Reservoir Outflow	Interim Ramping Rate	Proposed Ramping Rate
0 to 250 cfs	125 cfs/hr	25 cfs/hr
250 to 1,000 cfs	250 cfs/hr	250 cfs/hr
>1,000 cfs	1,000 cfs/hr	1,000 cfs/hr

The maximum ramping rates at release levels below 1,000 cfs differ from authorized rates. However, every effort is made to comply with the interim rates (USACE 1998a,b). These ramping rates are intended for flood control activities only. Flow changes above 1,000 cfs release are generally limited to a rate of 1,000 cfs/hr to protect against bank sloughing and are not related to fish stranding issues. Lower ramping rates at lower reservoir flow releases are to protect against fish stranding. The ramping rate guidelines are followed for flood operations that ramp flows down as well as releases that ramp flows up (Bond, USACE, pers. comm.).

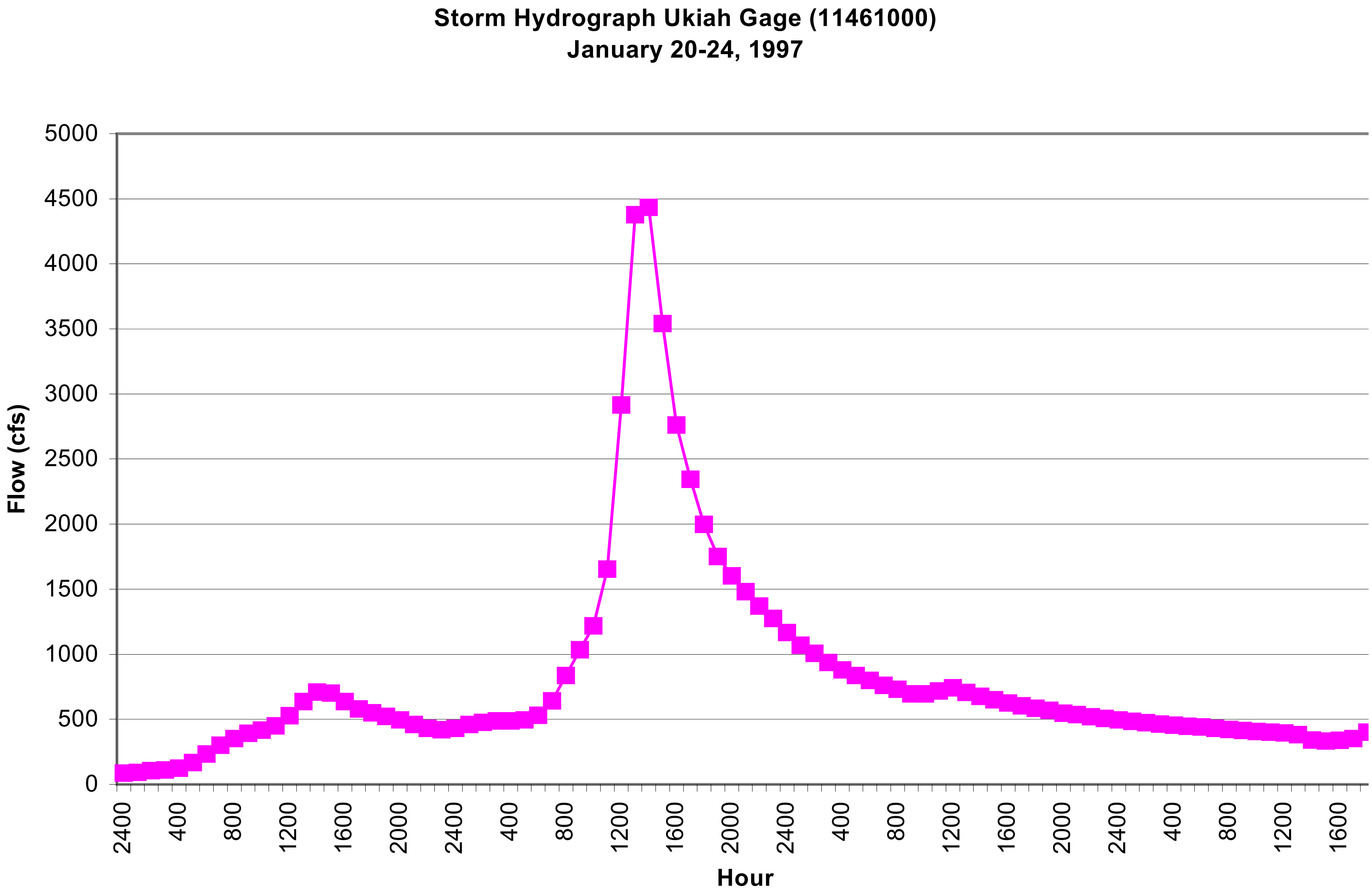

Figure C-1 Storm Hydrograph Ukiah Gage

It is unlikely that ramping-up rates associated with flood control operations would have an effect on listed species. Dam releases during flood control operations are made when downstream tributary flows are receding after a storm event, thereby reducing rather than augmenting natural flood peaks. Ramping-up rates follow the interim guidelines, so that when release flows are above 1,000 cfs, ramping occurs at no more than 1,000 cfs/hr.

This ramping rate is lower than natural flow increases associated with storm events. The USGS gage at Ukiah (11461000) located above the Forks was inspected and evaluated for natural flow changes for the period November 1995 to June 1999. Flows at the Ukiah gage are not regulated, and therefore represent natural flow fluctuations. On the rising limb of the storm hydrograph, hourly increases in flows above 1,500 cfs average 390 cfs/hr, and 10 percent of the time (90^{th} percentile) exceed 960 cfs/hr. A storm hydrograph for January 20 to 24, 1997 is shown in Figure C-1. From USGS stage data for this station, the maximum stage change associated with the rising limb of this storm event is approximately 1.9 feet/hr. The stage change associated with the average 390 cfs/hr increase in flows is approximately 0.5 foot (when flows are greater than 1,000 cfs). These data indicate that natural stage changes are sometimes greater than the Hunter criteria.

C.1.7.2 DAM INSPECTION AND MAINTENANCE

In addition to ramping during flood control operations, change in flow releases from Warm Springs Dam and Coyote Valley Dam are scheduled annually for dam maintenance and inspection activities. To perform the annual and periodic dam inspection and maintenance work, ramping-down flow releases is necessary for conduit inspections. Ramping rates during dam inspection and maintenance have in recent years been determined by consultation between USACE and NOAA Fisheries prior to each year's annual inspection.

In addition to regular pre-flood inspection and maintenance activities, both dams have historically required infrequent but important testing of the outlet works to verify safe operation of the projects. Testing may include investigations to determine damages, identify the cause of damages, and verify the reliability of outlet works and changes in Standard Operating Procedures to insure the continued operational integrity of the project. The flow releases necessary for testing are not the same as those required for pre-flood inspection and maintenance activities. Testing flow releases is variable, and the need to conduct testing may arise at anytime throughout the year. An example of dam safety testing is the vibration analysis conducted in January, February, and March 1998 at Warm Springs Dam, where outflow varied between 50 cfs and 3,000 cfs. This testing was performed to investigate the reliability of the outlet works and to insure the continued safe operation of the dam.

C.1.7.3 RAMPING RATE EVALUATION CRITERIA FOR WARM SPRINGS AND COYOTE VALLEY DAMS

Evaluation criteria were developed to assess effects of the interim and proposed ramping rates. It is unlikely that ramping-up rates associated with flood control operations would affect listed species, so evaluation criteria were not developed for ramping up.

C.1.7.3.1 Ramping Release Rate 1,000 cfs to 250 cfs

Ramping may occur at higher or lower streamflow conditions during the winter and spring runoff periods as part of flood control operations. When the reservoir release is between 1,000 cfs to 250 cfs, the guideline for the ramping rate is 250 cfs/hr.

Evaluation criteria and scoring for ramping in the 1,000 cfs to 250 cfs flow range (Table C-36) are based on Hunter's (1992) guidelines and the interim ramping rates established by USACE in consultation with NOAA Fisheries and CDFG. The highest score is given if stage changes meet Hunter's (1992) guidelines, 0.16 foot/hr during periods when juveniles are present. Ramping that exceeds Hunter's (1992) guidelines by up to 100 percent, receive a score of 4. Ramping activities that exceed Hunter's guidelines by more than 100 percent but still meet the established ramping rate (250 cfs/hr) receive a score of 3. Ramping rates that exceed the interim flow criteria by up to 50 percent (i.e., up to 375 cfs/hr) receive a score of 2, and if ramping rates exceed the interim flow criteria by more than 50 percent (greater than 375 cfs/hr), the score is 1.

Table C-36 Ramping Evaluation Criteria for Streamflows 1,000 cfs to 250 cfs

Category Score	Evaluation Criteria Category
5	Meets 0.16-foot maximum stage change.
4	Within 100% of 0.16-foot criterion (0.32 foot/hr) for stage change.
3	Meets 250 cfs/hr ramping criterion.
2	Exceeds ramping criteria up to 50% (375 cfs/hr).
1	Exceeds ramping criteria by greater than 50% (>375 cfs/hr).

To determine if the ramping rates meet, or the extent to which they exceed the criteria in Table C-36, stage-discharge relationships were obtained from HEC-RAS modeling for the appropriate cross-sections. The HEC-RAS model provides information on the change in stage (depth) associated with a change in discharge. The model itself does not account for the effects of attenuation of releases by flow contributions from downstream tributaries or accretion in baseflow. Therefore, the HEC-RAS model may overestimate changes in stage for progressively downstream cross-sections. Pools, side-channels, and gravel bars attenuate the ramping rate by storing water from higher flows and releasing the water gradually. The largest actual changes in stage are expected closest to the dam.

On Dry Creek, the ramping evaluation includes a 1.5-mile-long reach below Warm Springs Dam. Ten cross-sections (103 to 112) were used in the assessment. On the mainstem Russian River, four cross-sections (48, 48.1, 49, 49.1) closest to Coyote Valley Dam, from approximately 3 miles to 5 miles downstream of the dam, were used. There are no cross-sections available for the East Fork Russian River (cross-section data were collected at two locations on the East Fork near Coyote Valley Dam in May 2000 by SCWA, but these cross-sections have not been used in the HEC-RAS modeling). Therefore, an evaluation of stage-discharge relationships relative to Hunter's guidelines could not be performed. However, flow release data at both dams were examined from recent years (1997 to 1999) to determine the extent to which flood control operations may be meeting the interim ramping criteria as designated in Table C-36.

C.1.7.3.2 Ramping Release Rate 250 cfs to 0 cfs

Ramping of release flows in the range of 250 cfs to 0 cfs typically take place in winter or spring as flood control operations reduce flows from much higher rates following storm events. Flows at the Ukiah gage, above the Forks on the mainstem Russian River, are usually greater than 500 cfs when flood control operations are ramping at release rates less than 250 cfs. During most of the year, juvenile salmonids are expected to be present, and therefore the criteria for juveniles applies (0.16 foot/hr). The evaluation criteria (Table C-37) are similar to that presented for the release rates 1,000 cfs to 250 cfs, except that the interim flow guidelines call for a maximum ramping rate of 125 cfs/hr when reservoir releases are within the 250 cfs to 0 cfs range (USACE 1998a,b). Proposed maximum ramping rates for this flow range is 25 cfs/hr.

Table C-37 Ramping Evaluation Criteria for Streamflows 250 cfs to 0 cfs

Category Score	Evaluation Criteria Category
5	Meets 0.16 foot/hr maximum stage change
4	Within 100% of 0.16-foot/hr criterion (0.32 foot/hr) for stage change
3	Meets 125 cfs/hr ramping criterion
2	Exceeds 125 cfs/hr ramping criteria up to 50% (188 cfs/hr)
1	Exceeds 125 cfs/hr ramping criteria by greater than 50% (>188 cfs/hr)

The analysis procedure using the HEC-RAS model to determine change in stage at the designated cross-sections is exactly the same as that discussed for the 1,000 cfs to 250 cfs ramping range.

C.1.7.4 ANNUAL AND PERIODIC DAM INSPECTIONS AND MAINTENANCE

C.1.7.4.1 Issues of Concern

Annual and periodic pre-flood inspections take place at both Coyote Valley Dam and Warm Springs Dam. Inspections took place in September 1998 and June 1999. In 2000, dam inspection and maintenance activities took place during May. The 2000 inspection for Coyote Valley Dam was scheduled for May, but during the ramp-down steelhead fry stranding was noted downstream. The inspection was cancelled and performed in October (Eng 2000).

It is unlikely that maintenance inspections for Coyote Valley Dam will occur in the spring except for actions classified as emergency situations. Under baseline conditions, flows were reduced or completely shut down, usually for periods of several hours, to accomplish the inspections. Additionally, flows were reduced or shut down to perform periodic maintenance activities on the dams. Depending on the maintenance activities to be performed, flows were reduced or shut down for periods lasting several hours to 1 day or longer. Ramping rates and reduced streamflow conditions are the two primary issues of concern associated with annual and periodic dam inspections and maintenance.

C.1.7.4.2 Ramping Rates

To perform the annual and periodic dam inspection and maintenance work, ramping-down flow releases is often necessary. In recent years, ramping rates have been determined by consultation between USACE and NOAA Fisheries before each year's annual inspection. In the past, stranding has been documented below Coyote Valley Dam, but not below Warm Springs Dam. These cases are discussed in the next section. At Warm Springs Dam, the ramping rate is typically 25 cfs/hr. At Coyote Valley Dam, the typical ramping rate during inspection activities is 50 cfs/hr. However, at Coyote Valley Dam, the USACE ramp-down rates are done at the smallest increments possible. For the 2001 and 2002 inspections, rates were 30 to 50 cfs for Coyote Valley Dam (ENG 2001; 2002).

Issues of concern relative to ramping rates during pre-flood inspection and maintenance activities are primarily related to stranding and dewatering. Depending on when maintenance and inspection activities take place, ramping may affect both fry and juvenile life-history stages.

C.1.7.4.3 Reduced Streamflows During Inspection and Maintenance

During shut-down or reduction of flow from either dam, stranding and mortality may occur, particularly for fry. A bypass flow of approximately 25 cfs to 28 cfs is usually maintained at Warm Springs Dam during pre-flood inspections. During inspections at Coyote Valley Dam, there is no bypass capability, so flow releases must be completely shut down. However, a small flow is maintained below the dam for up to several hours as the plunge pool and afterbay drain, or if the stilling basin is dewatered for inspection as occurred in June 1999. In 2000, maintenance activities were scheduled in May with the hope that higher streamflows in the mainstem would attenuate effects from flow reductions at Coyote Valley Dam.

Flow contributions on the mainstem below the Forks is always greater in the spring compared with the summer or fall months. Inspection of flow records since 1995 at the Ukiah gage (USGS gage 11461000), located above the Forks, indicates that flow has never been less than 11 cfs. Flows are usually greater than 35 cfs, and may be up to several hundred cfs. In contrast, flows in September at the Ukiah gage are almost always 1 cfs to 2 cfs. Streamflows on the East Fork during maintenance activities were expected to be very low since there was no bypass capability. However, observations during the June 1999 inspection and maintenance indicate that some water depth was maintained in the pools, and a small flow was apparent (although it was not measured) despite flow reductions to 0 cfs at Coyote Valley Dam (Terry Marks, USACE, pers. comm. 2000).

Stranding due to ramping rates or partial dewatering of the channel during scheduled activities at Coyote Valley Dam occurred on the mainstem Russian River below the Forks when maintenance and pre-flood inspections were scheduled in May 2000. With the first decrease in flows from approximately 168 cfs to 118 cfs, over ten salmonids were stranded below the Forks, and the decision was made to abandon the scheduled maintenance at that time (T. Daugherty, NOAA Fisheries pers. comm. 2000). During a scheduled maintenance activity on Dry Creek in May 2000, only a few (eight) steelhead were found stranded by the time the ramp-down was completed (R. Sundermeyer, ENTRIX, and T. Daugherty, NOAA Fisheries, pers. comm. 2000).

In October 1997, the emergency water supply pipeline at Warm Springs Dam was repaired and the annual pre-flood inspection performed. A minimum 28 cfs release was maintained from the dam for periods lasting for approximately 8 hours over several days in order to perform the repairs and inspection. Dry Creek was monitored by USACE during this time. The monitoring concluded that there was adequate flow for juvenile salmonids, since no mortalities or stranding were discovered (USACE 1997).

A periodic inspection was conducted at Coyote Valley Dam on September 9, 1998. There were no bypass flows during this inspection. Streamflow was monitored 4 miles downstream from the dam, but flow velocities were too low to measure with a current meter. Discharge was estimated to be less than 30 cfs. Further downstream at Hopland, the USGS gage indicated the discharge was below the rating table (indicating less than 200 cfs) for approximately 7 hours. Some juvenile steelhead were stranded and rescued below the dam on the East Fork to approximately 12,000 feet downstream on the mainstem Russian River below the Forks.

A pre-flood inspection at Coyote Valley Dam was performed on June 10, 1999. Approximately 10 hours were planned to conduct the inspection of the outlet works conduit and stilling basin, but this was cut short by a few hours. Releases from Coyote Valley Dam were below the minimum 25 cfs instream flow requirement for approximately 4 hours. SCWA petitioned the SWRCB for a temporary urgency change in minimum flow requirements, which was approved for this inspection. During the inspection, streamflow at the Ukiah gage (above the Forks) was 12 cfs to 14 cfs, and at Hopland it ranged between 93 cfs to 221 cfs. Although water was pumped out of the stilling basin (contributing approximately 5 cfs downstream), the stilling basin was never dewatered and an inspection of it was cancelled. Direct mortality was a concern due to potential entrainment when pumping the stilling basin. NOAA Fisheries issued an Incidental Take Statement in the Biological Opinion for the maintenance activity and required monitoring of the East Fork Russian River for strandings and temperature (NMFS 1999). No strandings or fish mortalities were found, and there were no significant increases in temperature (Terry Marks, USACE, pers. comm. 2000). During work scheduled at Coyote Valley Dam in October of 2000, as well as September of 2001 and 2002, no stranding or mortality was documented.

C.1.7.4.4 Evaluation Criteria for Ramping During Dam Maintenance and Inspections

Evaluation criteria for ramping during pre-flood inspections are based on the historical incidence of stranding that has been documented at Warm Springs Dam and Coyote Valley Dam. At Coyote Valley Dam, flow reductions of 50 cfs/hr during May have resulted in fish stranding on the mainstem Russian River. Stranding of juvenile steelhead was documented in September 1998 on the lower East Fork and mainstem Russian River.

At Warm Springs Dam, flow reductions of 25 cfs/hr have resulted in very limited stranding of steelhead during May when fry are present. Stream widths in Dry Creek and the upper mainstem below the Forks are similar (100-foot to 150-foot widths), and HEC-RAS modeling indicates similar stage-change relationships for 25 cfs/hr flow reductions. Therefore, one set of ramping criteria has been developed for application to both locations. Evaluation criteria for ramping rates are given in Table C-38. Scoring criteria distinguish times when fry are present and when only juveniles are present.

Table C-38 Evaluation Criteria for Low Reservoir Outflows (250 cfs to 0 cfs)[1] during Dam Maintenance and Pre-Flood Inspection Periods

Change in Flow (cfs/hr)	Score Juvenile	Score Fry
0 – 10	5	5
10 – 20	5	4
20 – 30	4	3
30 – 40	3	2
40 – 50	2	1
> 50	1	0

1 Only during maintenance activities do releases approach 0 cfs. Bypass flows of 25 cfs would be provided during maintenance.

The evaluation criteria in Table C-38 are appropriate only for those streamflow conditions when there is relatively low flow contribution below the Forks on the mainstem. Ramping during flood control operations occur when streamflows are much higher on the mainstem, typically 500 cfs or more at the Ukiah gage. Under these streamflow conditions, there are no exposed channel bars on the mainstem or the East Fork, and stranding is much less likely. Therefore, ramping evaluation criteria appropriate for assessing potential stranding are those previously defined in Table C-38 for flows between 250 cfs to 0 cfs.

Evaluation criteria listed in Table C-38 should be used when streamflows are less than 500 cfs at the Ukiah gage. Stage-discharge relationship information generated by the HEC-RAS model was used as an independent check to identify how the flow ranges in the evaluation criteria compare with Hunter's criteria. Stage changes associated with 25 cfs/hr reductions at Warm Springs Dam and both 50 cfs/hr and 25 cfs/hr at Coyote Valley Dam were modeled.

Warm Springs Dam

Stage changes associated with 25 cfs/hr incremental flow reductions beginning at 250 cfs, then 225 cfs, 200 cfs, 175 cfs, 150 cfs, 125 cfs, 100 cfs, and 75 cfs are shown for ten cross-sections on Dry Creek in Figure C-2. The change in stage associated with a given streamflow is shown by the height of each bar. For example, the bar on the x-axis at 250 cfs for cross-section 103 represents a flow reduction from 250 cfs to 225 cfs, and the associated stage change is indicated on the y-axis as less than 0.08 foot. A bar for cross-section 103 representing a flow reduction from 225 to 200 cfs, and the associated stage change indicated on the y-axis is a little greater than the change at 250 to 225 cfs. Transect 112 is closest to Warm Springs Dam, and transect 103 is the most distant. Overall, the stage change associated with 25 cfs/hr ramping meets the 0.16 foot/hr Hunter criteria within most of the flow ranges below 250 cfs.

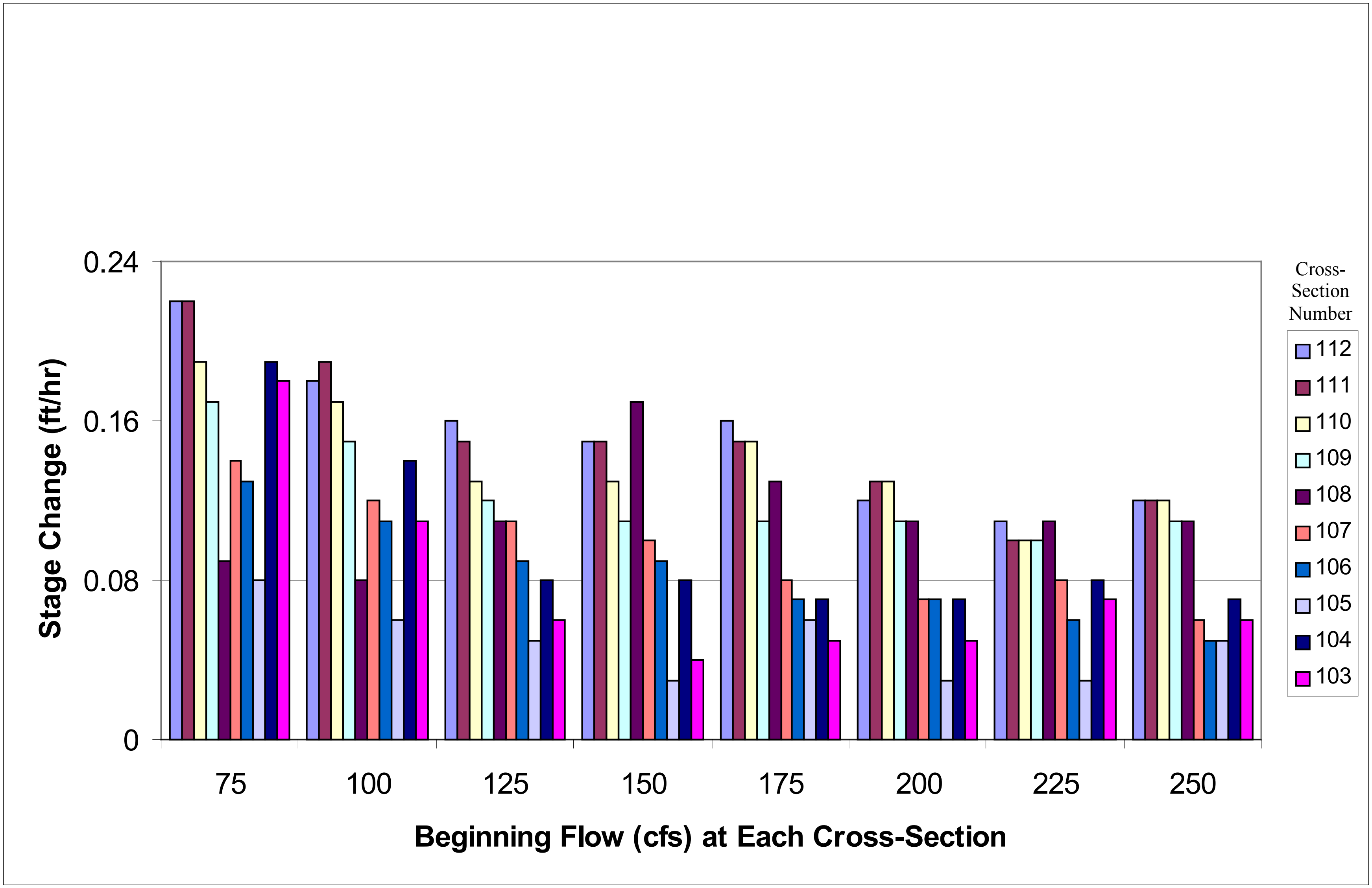

Figure C-2 Stage Changes Associated with 25 cfs/hr Ramping Rate at Warm Springs Dam

Russian River and East Fork below Coyote Valley Dam

Stage changes associated with 25 cfs/hr incremental flow reductions beginning at 250 cfs, then 225 cfs, 200 cfs, 175 cfs, 150 cfs, 125 cfs, 100 cfs, 75 cfs, and 50 cfs are shown for four cross-sections on the mainstem Russian River below the Forks in Figure C-3. Cross-section 49.1 is closest to the dam (approximately 2.5 miles downstream), and cross-section 48 is furthest from the dam (approximately 5 miles downstream). The change in stage associated with a given streamflow is shown by the height of each bar. Cross-section data for the East Fork Russian River have recently been obtained, but stage discharge relationships from HEC-RAS modeling were not developed for this analysis. Stage changes associated with 50 cfs/hr incremental flow reductions are shown in Figure C-4.

At 25 cfs/hr reductions, the 0.16 foot/hr criterion is met at most flow intervals in all four of the cross-sections for flow ranges below 250 cfs. At 50 cfs/hr reductions, the 0.16 foot/hr criterion is exceeded, and stage changes are generally in the range of 0.24 to 0.32. This suggests that the potential for stranding is greater at Coyote Valley Dam.

C.1.7.5 INFLATABLE DAM

An inflatable dam is deployed on the Russian River upstream of the Mirabel area. The dam is raised when river flows are declining (generally in the spring) and is lowered when river flows are rising (approximately once or twice per year). Flow recessions during dam inflation or deflation have the potential to result in juvenile fish stranding. Three sets of evaluation criteria were developed based on rate of stage change, habitat features, and frequency of flow reductions.

Hunter's criteria, as well observations on fish stranding downstream of Coyote Valley Dam and Warm Springs Dam, were used to develop criteria to evaluate the effects of the rate of stage change. During inspection and maintenance activities scheduled in the spring at Coyote Valley Dam, 50 cfs/hr reductions in flow have resulted in significant stranding of juvenile fish in the mainstem below the Forks. Results of HEC-RAS hydraulic modeling of the mainstem Russian River below the Forks presented in *Interim Report 1* (ENTRIX, Inc. 2000) indicate that 25 cfs/hr flow reductions would result in stage changes that meet Hunter's 0.16 foot/hr stage-change criterion for most flow intervals. Only limited stranding of fry occurred in Dry Creek (May 2000) when releases from Warm Springs Dam were ramped down at a rate of 25 cfs/hr. Stage-discharge relationship information generated by the HEC-RAS model on cross-sections below Dry Creek indicates that the ramping rate of 25 cfs/hr meets the 0.16 foot/hr Hunter criterion within most of the flow ranges below 250 cfs (ENTRIX, Inc. 2000).

These observations on fish stranding and results of hydraulic modeling suggest that the Hunter 0.16 foot/hr stage-change criterion may be protective for juvenile salmon in the Russian River watershed. Because fry are more vulnerable to flow recessions than other life-history stages of salmon, a more stringent evaluation criterion of 0.08 foot/hr is applied.

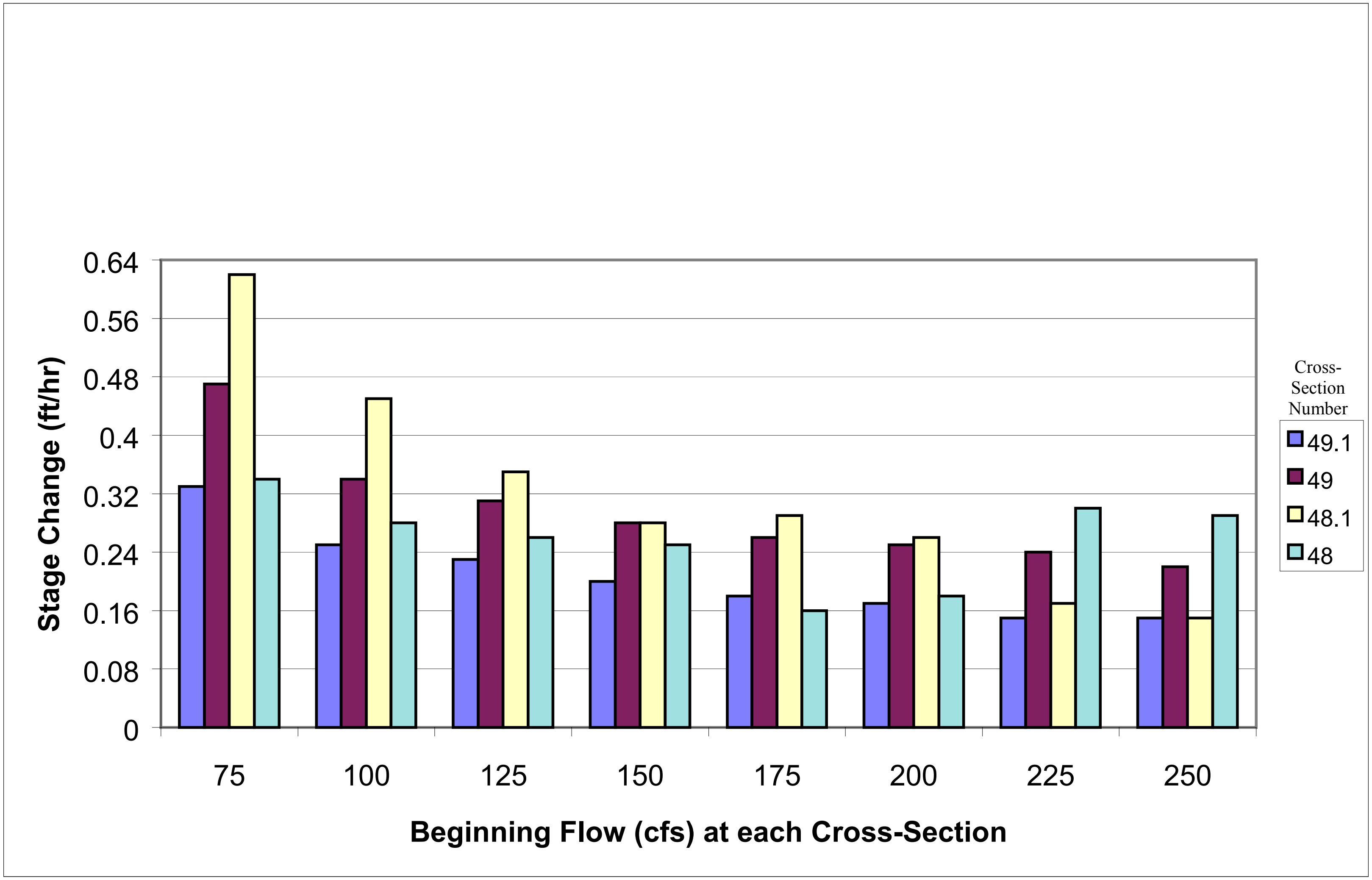

Figure C-3 Stage Changes Associated with 25 cfs/hr Ramping Rate on Mainstem Russian River below Coyote Valley Dam

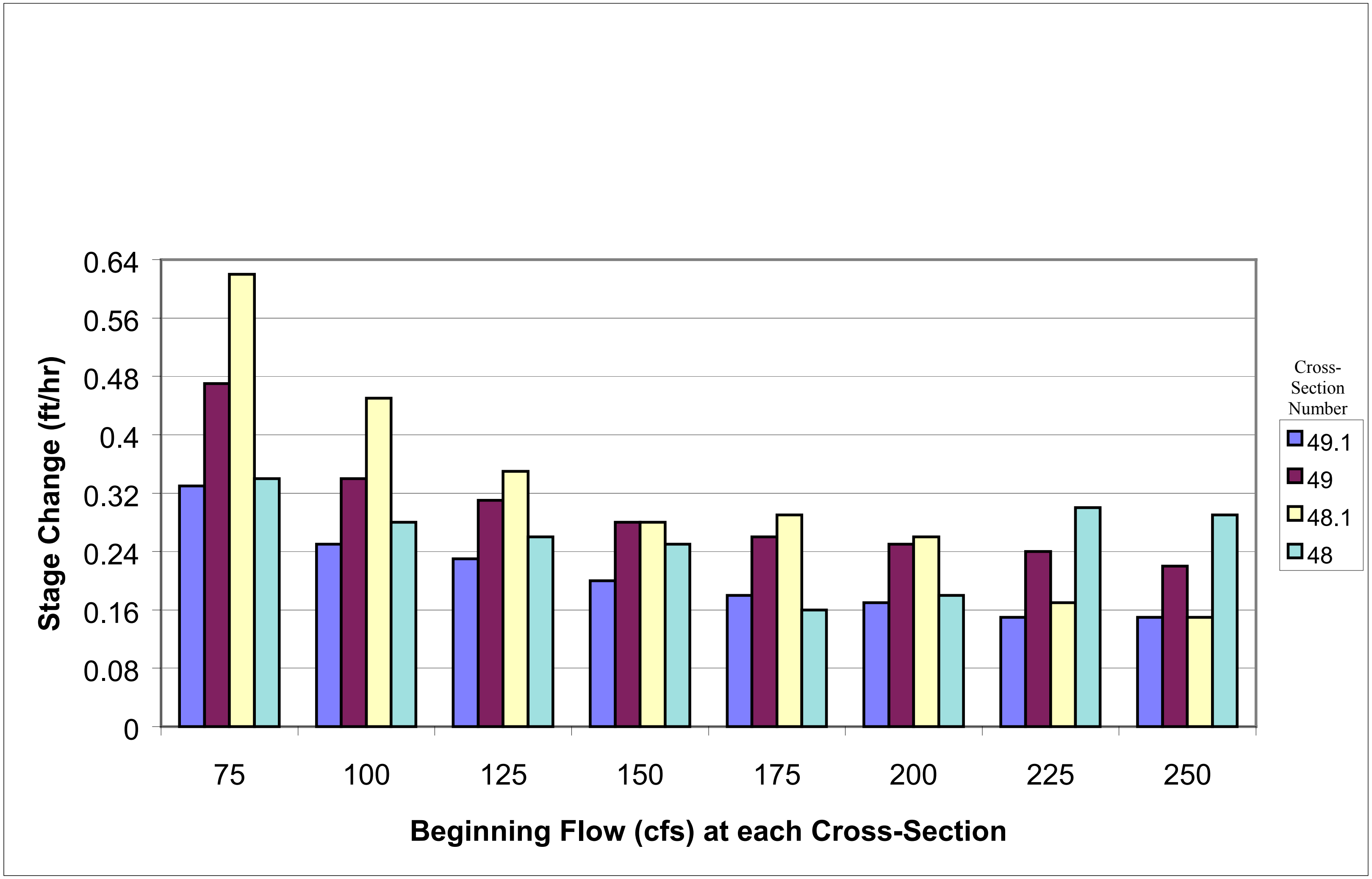

Figure C-4 Stage Changes Associated with 50 cfs/hr Reductions in Flow at Cross-Sections in Mainstem Russian River below Coyote Valley Dam

USACE, in consultation with NOAA Fisheries and CDFG, developed interim guidelines for release changes during flood control operations to protect spawning gravel and juvenile salmonids in Dry Creek and the Russian River. These criteria are less stringent than the Hunter criteria, but are appropriate to use when river flows are high. The dam is generally inflated or deflated when rising river flows are expected. These criteria are therefore appropriate for intermediate scores for this evaluation when the inflatable dam is inflated or deflated. The channel of the mainstem Russian River at Mirabel is larger than it is near Coyote Valley Dam or in Dry Creek, and therefore an equivalent flow change at Mirabel would result in a smaller stage change in this river reach.

At high river flows, the river is generally wetted from bank to bank and therefore the risk of stranding would be low. When flows are lower, riffles may be subject to dewatering and side pools may become isolated. Therefore the risk of stranding would be higher. A conservative estimate of flows below which dewatering may occur is 500 cfs. Stage changes below 500 cfs should be lower than above that level to protect salmonids.

USACE criteria designate three ramping rates determined by river flow. This evaluation uses the mid-flow category, which is for a river flow between 250 cfs and 1,000 cfs. The ramping criterion for these flow conditions is a maximum rate change of 250 cfs/hour. To estimate the stage change that is related to the interim ramping criteria at the inflatable dam, two cross-sections in the impounded area are used to correlate change in water elevation to a 250-cfs flow difference. Using HEC-RAS modeling, a 250-cfs flow change behind the inflatable dam is estimated at approximately a 0.32-foot stage-change.

Evaluation criteria were developed to address the rate of stage change according to lifestage. For juvenile salmonids, the Hunter criterion of 0.16 foot/hr is given a score of 5 (Table C-39). A score of 4, 3, and 2 are assigned to stage-change rates of the inflatable dam of 0.32 foot/hr, 0.48 foot/hr, and 1.4 feet/hr, respectively (based on USACE criteria). Because fry are more vulnerable, evaluation criteria for fry are more stringent than for juveniles. For fry, a score of 5 is assigned to stage-change rates of 0.08 foot/hr (Table C-40). A similar reduction in stage-change rate is used to assign the remaining score criteria.

Habitat features can affect stranding during flow recessions. The risk of stranding for a given reduction in flow increases with low-gradient river channel configuration, presence of long side channels, larger substrate type, and frequency of flow reductions. A river channel with many side channels, potholes, and low-gradient gravel bars has a greater incidence of stranding than a river confined to a single channel with steep banks (Bauersfeld 1978, Beck Associates 1989, and Hunter 1992). Most documented observations of stranding have occurred on gravel and vegetation (Becker et al. 1981 and Satterthwaite 1987).

Evaluation criteria were developed for habitat features that affect the risk of stranding during flow recessions. A single, steep-sided channel with fine substrate and no instream vegetation or potholes is likely to present no risk. The presence of side channels, low-gradient banks, gravel bars, potholes, or instream vegetation would increase the risk. A large area with many habitat features that are likely to induce stranding would have a greater risk than a smaller area with fewer of these habitat features. A score of 5 is assigned if local habitat conditions are unlikely to induce stranding during flow recessions. A score of 0 is given if more than 30 percent of the area of the habitat contain features that are likely to induce stranding (Table C-41).

Salmonids may have developed local adaptations to the frequency of naturally occurring flow recessions. However, an increase in the frequency of flow-reduction events provides additional opportunities for stranding or displacement of juvenile salmonids. Frequent fluctuations, such as daily fluctuation associated with hydroelectric project peaking operations in other river basins, provide much more opportunity for stranding than an occasional event such as a flow reduction after a seasonal flood. A score of 5 is assigned when the frequency of flow reductions due to deflation of the inflatable dam is less than two times a year. A score of 0 is assigned when reductions in flow occur daily (Table C-42).

Table C-39 Stage-Change Evaluation Criteria for Dam Inflation and Deflation for Juvenile and Adult Salmonids

Category Score	Evaluation Criteria Category
5	Meets 0.16 foot/hr maximum stage change.
4	Meets 0.32 foot/hr maximum stage change.
3	Meets 0.48 foot/hr maximum stage change.
2	Meets 1.4 feet/hr maximum stage change.
1	Greater than 1.4 feet/hr maximum stage change.

Table C-40 Stage Change Evaluation Criteria for Dam Inflation and Deflation for Fry

Category Score	Evaluation Criteria Category
5	Meets 0.08 foot/hr maximum stage change.
4	Meets 0.16 foot/hr maximum stage change.
3	Meets 0.32 foot/hr maximum stage change.
2	Meets 0.48 foot/hr maximum stage change.
1	Greater than 0.48 foot/hr maximum stage change.

Table C-41 Habitat/Flow Recession Interaction Evaluation Criteria for Fry, Juvenile, and Adult Salmonids

Category Score	Evaluation Criteria Category
5	Habitat features unlikely to induce stranding.
4	Few habitat features present to induce stranding.
3	Some habitat features that induce stranding, but area affected is small (<30%).
2	Many habitat features that induce stranding, but area affected is small (<30%).
1	Some habitat features that induce stranding, area affected is large (>30%).
0	Many habitat features that induce stranding, area affected is large (>30%).

Table C-42 Flow-Reduction Frequency Evaluation Criteria for Fry, Juvenile, and Adult Salmonids

Category Score	Evaluation Criteria Category
5	Less than 2 fluctuations per year in critical habitat.
4	Between 3 and 9 fluctuations per year in critical habitat.
3	Between 10 and 29 fluctuations per year in critical habitat.
2	Between 30 and 100 fluctuations per year in critical habitat.
1	More than 100 fluctuations per year in critical habitat.
0	Daily fluctuations in critical habitat.

C.1.8 Criteria for Construction, Maintenance, and Operation Activities

Direct effects from construction, operation, and maintenance activities may occur during water supply, channel maintenance, or restoration activities. These effects include fine sediment input to the stream, short-term increased turbidity, and direct injury or mortality of fish. Vegetation removal activities may have immediate and direct effects associated with the use of herbicides or mechanical removal methods, and indirect effects to habitat related to changes in instream habitat or the amount and quality of the riparian corridor. Channel maintenance activities such as streambank and streambed stabilization, sediment maintenance, debris removal, and vegetation control have potential long-term effects on salmonid habitat.

C.1.8.1 Fine Sediment and Turbidity

Activities that take place within a stream or on the streambanks may increase sediment input to the stream. By implementing effective best management practices (BMPs) during construction or maintenance activities, effects may be minimized.

Maintenance or construction activities can affect salmonids or their habitat in the immediate work area or in nearby areas downstream of the activity. If activities take place when no life-history stage for the species is present, then no negative short-term effect would be expected. If a construction or maintenance activity takes place during the low-flow period in the summer and fall seasons, potential direct, short-term effects would be restricted to juvenile salmonids and their rearing habitat, and some limited steelhead and Chinook salmon migration.

Evaluation criteria for sediment control address two components: instream and upslope sediment control (Table C-43). For the first component, instream sediment control, a high score indicates instream work practices with the highest degree of sediment containment, and a low score indicates poor or no sediment containment measures. Working in a stream that is dry receives a score of 5. Rerouting streamflow from the construction area into a clean bypass, or other method that reroutes streamflow, isolates the construction area and prevents sediment input to the stream; therefore, these options are given a fairly high score of 4. A clean bypass is routing streamflow around the maintenance activity so that continuity of flow and water quality is maintained downstream. A clean bypass isolates the work area from the wetted stream channel. For instream work in a wetted channel that does not use a bypass, there is a greater potential for sedimentation downstream, unless other effective methods of controlling sedimentation are used. For example, SCWA typically uses a gravel berm downstream to filter turbid waters and reduce potential sedimentation. Such effective sediment control measures are given a moderate score of 3. Limited sediment control is a measure that is only partially effective, and that may allow significant turbidity and sedimentation. Limited sediment control measures receive a score of 2, and no instream sediment control measures in wetted channels receive the lowest score of 1.

A second component of sediment control is identified as upslope sediment control. Depending on the site-specific characteristics, upslope sediment control may include either streambanks that are immediately adjacent to the channel, or in some cases, may include more distant upland areas where erosion control measures are employed. This component evaluates the amount of disturbance, the effectiveness of erosion control measures, and whether bank stabilization is improved or degraded. Similar to the instream component, a high score indicates minimal or no slope disturbance and a low score indicates maintenance activities that are likely to cause slope failure or bank erosion, with resulting sediment input.

Table C-43 Sediment Containment Evaluation Criteria

Category Score	Evaluation Criteria Category
	Component 1: Instream Sediment Control
5	Project area does not require rerouting streamflow.
4	Clean bypass or similar method used.
3	Effective instream sediment control (e.g., berm/fence).
2	Limited sediment control.
1	No instream sediment control.

Table C-43 Sediment Containment Evaluation Criteria (Continued)

Category Score	Evaluation Criteria Category
	Component 2: Upslope Sediment Control
5	No upslope disturbance, or an increase in upslope stability.
4	Limited disturbance with effective erosion control measures.
3	Moderate to high level of disturbance with effective erosion control measures.
2	Action likely to increase sediment input into stream.
1	Action likely to result in slope failure, bank erosion, an uncontrolled sediment input to the channel, or major changes in channel morphology.

C.1.8.2 EVALUATION CRITERIA FOR INJURY TO FISH

Work in a streambed that has flowing water or standing pools may result in direct injury or mortality to fish or incubating eggs. Furthermore, displaced fish may be subjected to short-term stress, predation, or competition.

Immediate effects from construction or maintenance activities are scored according to the opportunity for injury to protected species (Table C-44). BMPs are generally implemented to reduce the risk of injury to fish and may include scheduling the work when protected species are not present or when the stream channel is dry, conducting a biological survey of the project area to assess appropriate BMPs, isolating the project area from streamflow, and providing escape or rescue for fish that may be present. Site-specific factors dictate appropriate BMPs. For example, isolating a construction or maintenance area from streamflow may be a preferred alternative for some projects. However, this may result in an unacceptable disruption of habitat for other activities, such as activities that take place in a long reach of stream but involve minimal instream work. While a fish rescue may reduce the risk of injury, it has its own risks associated with it, and there may be times when providing escape is a preferred alternative.

High scores are associated with activities that have a low risk of injury, such as those that do not take place in the channel or that take place in a dry channel. Some activities require almost no interaction with the stream channel or water in the stream. These include maintenance activities related to road maintenance and scour holes around culverts. If activities take place when no fish species are present, then no direct injury to fish would be expected. The greater the interaction with the stream, the higher the risk of direct mortality to fish and effects associated with increased turbidity and sedimentation of aquatic habitat. Occasionally, a project may require equipment in the flowing channel. Appropriate BMPs, such as project area surveys by a qualified biologist, isolation of the project area from flow, and fish rescue or escape, reduce the potential for direct injury from equipment or due to stranding.

The lowest scores are given to activities that occur in a wetted channel where appropriate BMPs are not applied or applied in a limited way. There may be site-specific considerations that limit the ability of staff to apply appropriate BMPs. For example, emergency work after a landslide may restrict the ability of staff to implement all practices that might be desirable.

Table C-44 Opportunity for Injury Evaluation Criteria

Category Score	Evaluation Criteria Category
5	Project area is above the high-flow WSE defined by the 1.5 year bankfull event and/or above the tops of bars, and requires no isolation from flow.
4	Project area is within dry part of channel, or construction and maintenance activity scheduled when species of concern are not present.
3	Appropriate BMPs are applied; e.g., project area survey, escape, or rescue provided, project area isolated from flow (if appropriate).
2	Limited ability to apply appropriate BMPs.
1	Appropriate BMPs are not applied.

The risk to protected fish species may be greater if there are sensitive biological or habitat conditions in a particular area. For example, if a maintenance activity is scheduled in the late summer in the upper mainstem Russian River, where important rearing habitat is known to occur, the effects may be more significant than if the work were performed in the Mirabel area where high summer water temperatures may limit the number of listed fish species present. The level of risk is qualified and described where there is a general knowledge of the tributary or channel reach conditions where the work is performed.

C.1.8.3 Direct Effects of Vegetation Control

Vegetation may be removed from streambanks and stream-channel bottoms to maximize channel flow capacity and to reduce the risk of fires. Non-native vegetation removal may be conducted as part of a restoration action. Herbicides can have direct effects on fish habitat.

Other vegetation control methods such as hand-trimming or mechanized mowing are primarily related to indirect, long-term habitat alteration effects (although immediate effects may also occur upon implementation). The indirect effects associated with vegetation maintenance activities are discussed in Section C.1.8.4.

C.1.8.3.1 Direct Effects Related to Herbicide Use

Spraying herbicides to control vegetation in channels can have an immediate direct effect on fish and water quality. Herbicides have been developed to minimize effects in riparian and wetland habitats. For some plants, such as the highly invasive, non-native weed *Arundo donax* (Giant Reed), a combination of mechanical/hand-clearing and herbicide use are effective, while the use of one or the other alone is not. An herbicide approved for aquatic use should be used near streams. For example, a commonly used herbicide that has been approved by the EPA for use near aquatic areas is glyphosate (Rodeo®).

Glyphosate, when used according to directions, is practically nontoxic to fish and may be slightly toxic to aquatic invertebrates (EXTOXNET 1996).

C.1.8.3.2 Evaluation Criteria for Vegetation Control Associated with Herbicide Use

Vegetation control evaluation criteria (Table C-45) assess the amount and quality of chemicals released into the aquatic environment when herbicides are used. Higher scores are associated with practices that use only an aquatic contact herbicide, and limit herbicide use to smaller, targeted areas. Herbicide application can be limited with the use of an individual backpack unit as opposed to being broadcast over a wider area, or it can be applied over a large area with aerial spraying. Moderate to heavy herbicide use is associated with large-scale vegetation removal activities; for example, if a large infestation of *Arundo* had to be removed.

Table C-45 Evaluation Criteria for Vegetation Control Associated with Herbicide Use

Category Score	Evaluation Criteria Category
5	No chemical release.
4	Limited use of herbicide approved for aquatic use in riparian zones or over water.
3	Moderate to heavy use of herbicide approved for aquatic use in riparian zones or over water.
2	Use of herbicide not consistent with instructions.
1	Use of herbicide not approved for aquatic use in riparian zones or over water.

C.1.8.4 INDIRECT EFFECTS OF VEGETATION CONTROL

Vegetation control methods include removal by hand-trimming or mechanized mowing, or spraying, and indirectly by excavation of sediments and gravel bars. Another indirect method of vegetation control is to plant desirable native riparian vegetation that will exclude the establishment of non-native or undesired vegetation.

The duration of potential effects of vegetation removal may be short if vegetation grows back quickly, or long-term if vegetation is restored over a long time (for example, it takes years for trees to reestablish). However, effects of vegetation removal can be far more complex.

Riparian vegetation has several important functions for the quality of fish habitat (Meehan 1991). Water quality, including temperature and suspended sediment concentrations, may be influenced. Riparian vegetation, especially trees, provide canopy cover and shade, and removal may increase solar input and result in higher water temperatures in the summer. Loss of riparian vegetation may have a greater effect on temperature on narrow streams than on wide streams where the canopy covers only a small portion of the channel. Since salmonids occupy a wide variety of habitat types during various life-history stages, it is important to have quality habitat in small and large streams. On small streams, grasses and shrubs may be sufficient to provide beneficial effects, while on larger streams, shrubs and trees may be more effective.

Over the long-term, trees contribute to habitat diversity, often by creating high-quality pools or high-flow refuge habitat when they fall into the channel. This process of tree recruitment may help to control the slope and stability of the channel, particularly in forested regions (Beschta and Platts 1986). Instream vegetation such as willows can help stabilize gravel bars and provide high-flow refuge habitat. Streambank stability is also maintained and water quality improved by flexible vegetation such as willows and grasses. During floods, water transports large amounts of sediment in the stream. Vegetation mats on the streambank reduce water velocity, causing sediment to settle out and become part of the bank, increasing nutrients that are so important to productive riparian vegetation. Root systems of grasses and other plants can trap sediment to help rebuild damaged banks. Riparian vegetation provides cover, an important determinant of fish biomass. Well-sodded banks tend to gradually erode, creating undercuts important as refuge habitat.

Riparian vegetation provides a basis for food production. Vegetation provides habitat for terrestrial insects, which are an important food for salmonids. Plant matter provides organic material to the stream, essential for production of aquatic insects. This organic input is especially important to narrow, heavily shaded, headwater streams that support an aquatic insect community known as "shredders", which in turn supports salmonids. In sunnier, wider streams, an insect community known as "grazers" is supported by algal growth. Where cover and stream temperatures are not limiting, additional sunlight after limited vegetation removal may benefit primary productivity.

Vegetation removal can be beneficial if it involves the removal of non-native noxious species. Non-native vegetation, such as the invasive *Arundo donax*, can negatively alter critical habitat of salmonids, including alterations to the food web, the amount and quality of riparian and instream cover, streambank stability, and alterations to flow regimes. Replacement of non-native species with native species generally will help restore a naturally functioning, native, riparian ecosystem.

Separate evaluation criteria for vegetation control were developed for constructed flood control channels and natural channels. Constructed flood control channels are widened and straightened waterways that have been significantly altered and improved based on flood control criteria. The purpose of the improvements is to increase hydraulic capacity.

C.1.8.4.1 Vegetation Control and Removal in Constructed Flood Control Channels

Riparian vegetation is essential for building and maintaining stream structure and for buffering the stream from incoming sediments and pollutants. On natural channels when bank vegetation is reduced, flood events are more likely to accelerate changes in channel morphology such as widening or incision. However, constructed flood control channels are designed to be stable with minimal bank protection associated with riparian vegetation. As part of the design criteria, if flood velocities were calculated to exceed 6 fps, then hard-armoring was installed to protect sections of the bank from erosion

(SCWA 1983). Thus, under baseline conditions, removal of riparian vegetation on streambanks (except grass banks that are maintained) was anticipated to be an ongoing maintenance activity to preserve the channel design flood capacity.

Evaluation criteria for vegetation control are based on the extent of removal of native riparian vegetation (Table C-46). Higher scores are associated with activities that preserve or increase a riparian corridor composed of native species. Lower scores are given for maintenance practices that result in removal of riparian vegetation. The greater the extent of removal, the lower the score. Removal of invasive, non-native vegetation could have a beneficial effect because this may allow native riparian vegetation to establish.

For maintenance activities that include only selective removal of vegetation along access roads and between the access roads and fencelines, or for removal of non-native species, the highest score, 5, is given. The category score of 5 also includes “spot” or site-specific treatments that may require vegetation removal over very small distances, typically near structures such as culverts or at bridge crossings.

For maintenance activities that require more than selective removal of vegetation to keep access roads open, and thereby result in removal of vegetation across up to 25 percent of the cross-sectional area of the channel, the score is 4. When more than 25 percent and up to 50 percent of the cross-sectional area of vegetation is removed, then the score is 3. Removal of more than 50 percent and up to 75 percent of the vegetation represented in the cross-sectional area of the channel receives a score of 2, and more than 75 percent removal results in the lowest score, 1.

Table C-46 Vegetation Control Evaluation Criteria for Flood Control Channels

Category Score	Evaluation Criteria Category
5	No removal except selectively along access roads, fencelines, “spot” treatments, or to remove non-native species.
4	< 25% reduction in vegetation.
3	>25% to < 50% reduction in vegetation.
2	>50% to <75% reduction in vegetation.
1	>75% reduction in vegetation.

Consideration is also given to the life-history stage of listed species that are likely to be utilizing channels subject to vegetation maintenance and to the quality of habitat available. For example, if listed species are primarily using a flood control channel for migration rather than for rearing or spawning, the effect of vegetation loss is not considered to be as significant. If more than one life-history stage is potentially affected, the loss of vegetation becomes more significant. These considerations are addressed in conjunction with the scoring criteria listed above.

C.1.8.4.2 Vegetation Removal in Natural Channels

Constructed flood control channels are designed to be stable without the influence of riparian vegetation. Unlike the constructed flood control channels, riparian vegetation has an important effect on bank strength and stability in natural channels. Bank erosion and lateral channel migration contribute sediments to the stream if protective vegetation and root systems are removed from streambanks. Loss of vegetation decreases bedform roughness, thereby increasing velocities which may reduce the potential for sediment deposition on the channel margins or on the bank. Riparian vegetation provides cover, an important determinant of fish biomass. Additionally, well-sodded banks gradually erode, creating undercuts important as refuge habitat. Root systems of grasses and other plants can trap sediment to help rebuild damaged banks.

The potential for recruitment of trees and other large woody debris is probably much greater in natural channels compared with constructed flood control channels. This is due to the stable design and lack of lateral channel migration associated with the flood control channels. Meandering and lateral channel migration is often part of the natural channel processes that will cause bank erosion and tree recruitment. Therefore, removal of riparian vegetation in natural channels likely represents a greater loss of potential recruitment of large woody debris and resulting habitat diversity.

Natural channels tend to provide rearing and spawning habitat, in addition to migration, that most flood control channels do not provide. Therefore, vegetation maintenance activities in natural channels have a greater potential for altering habitat conditions that support multiple life-history stages. On this basis, greater weighting is given to habitat alteration effects resulting from vegetation removal in natural channels than in constructed flood control channels.

Evaluation criteria for vegetation control is similar to that for flood control channels, and is based on the extent of removal of native riparian vegetation (Table C-47). The scoring is slightly different in that there is no removal of vegetation associated with access roads or fencelines on natural channels (category score 5), and the percent of vegetation removal allotted within each of the scoring categories is lower than for flood control channels.

Table C-47 Vegetation Control Evaluation Criteria for Natural Channels

Category Score	Evaluation Criteria Category
5	No vegetation removal except “spot” treatment, or removal of only non-native species.
4	<10% reduction in vegetation.
3	>10% to <25% reduction in vegetation.
2	>25% to <50% reduction in vegetation.
1	>50% reduction in vegetation.

In conjunction with these scoring criteria, consideration is also given to the typical lengths of channel that are subject to vegetation maintenance. Maintenance practices that remove vegetation over long channel reaches are more likely to result in significant change to habitat conditions than shorter channel reaches. For example, complete removal of vegetation (100 percent) in the cross-sectional area over a 25-foot length of channel downstream of a culvert outfall (i.e., "spot" treatment) does not have the same degree of habitat-altering effects as 50 percent removal over a distance of 5,000 linear feet.

These criteria assess the amount of vegetation removed within a site. While limited vegetation removal in isolated sites may not negatively affect salmonid habitat, if the work is done over several sections of a stream and/or in prime spawning and rearing habitat, the net effect may be larger. For example, if willows are removed from several gravel bars to reduce streambank erosion in an important coho salmon stream, the net effect may be to significantly alter channel morphology, the amount of instream cover, and the availability of winter refugia from high flows. To avoid significant effects to salmonid habitat, vegetation removal in natural channels should be kept to a minimum and used only where there is an unacceptable threat from a 100-year-flood event or where a decrease in bank stabilization threatens a structure or property. Alternative solutions should be pursued where feasible. These include bioengineering practices to stabilize banks, tree planting to add bank stability and reduce understory growth, offsetting levees to increase floodplain, or digging floodplain level culverts to increase floodplain draining at culvert crossings.

C.1.8.5 STREAMBANK AND STREAMBED STABILIZATION

Activities are conducted to maintain existing bank stabilization structures in Dry Creek and the mainstem Russian River. Bank stabilization activities are proposed for the mainstem of the Russian River. Bank stabilization activities may also be conducted occasionally in natural channels in response to a catastrophic bank erosion event at the request of a landowner.

C.1.8.5.1 Maintenance of Streambank Stabilization Structures and Levees

Streambanks have been stabilized on Dry Creek and the mainstem Russian River using gravel revetments, steel jacks, sheet-pile, trees, and other materials. Levees are maintained to reduce bank erosion and flooding. On Dry Creek, concrete sills have been installed to provide grade control, preventing streambed incision and resulting accelerated streambank erosion. SCWA maintains these structures.

Potential effects related to maintenance of streambank stabilization projects may be both positive and negative. Positive effects are associated with reduction or prevention of erosion and resulting sedimentation in the channel. Negative effects may be associated with loss of riparian shading and increased water temperatures. Bank stabilization techniques may reduce the complexity of instream cover naturally provided by undercut banks, and exposed root wads. Additionally, the recruitment of spawning gravels, which are often supplied by natural bank erosion processes, may be impeded by bank stabilization structures. Streambed stabilization structures installed on Dry Creek are intended to reduce channel head-cutting and resulting streambank erosion.

Information is not available to quantify effects due to maintenance activities. Qualitative evaluation of these effects is based on the extent to which maintenance of bank stabilization structures reduce overstory canopy cover (shading), in-channel cover (undercut banks, exposed root wads, backwater areas), and gravel recruitment. The greater the loss of these habitat elements (in comparison with what normally could be supported by the stream reach), the greater the effect.

C.1.8.5.2 Bank Stabilization in the Russian River

Bank stabilization activities are conducted by the MCRRFCD and SCWA in the Russian River mainstem. Bank erosion occurs when flow is directed into the riverbank by large gravel bars that are often well-vegetated. Under the proposed project, gravel bars would be graded and overflow channels would be created at sites where the formation or growth of gravel bars is likely to cause bank erosion. The work would be conducted in site-specific areas to direct the river channel away from susceptible banks. Protocols would be implemented to reduce the potential for negative effects on salmonid habitat. This maintenance activity is closely linked to vegetation maintenance practices, which is also intended to ensure channel flood-capacity and to control bank erosion. The procedure for gravel bar grading and sediment removal conjunctively removes willows and other riparian vegetation from the bars. Since gravel bar grading is closely interrelated with removal of riparian vegetation growing on the bars, there is an associated loss of shade and canopy cover.

These bank stabilization activities in the Russian River are likely to affect the geomorphology of the channel. If stable bar development is prevented, the channel becomes straightened and sinuosity decreases. Decreased sinuosity reduces bank erosion, but also reduces the opportunity for pool development by limiting scour on the outside of meander bends. In addition, gravel bar grading generally results in a flatter streambed, reducing the hydraulic diversity and associated aquatic habitat diversity represented in the channel. This lack of hydraulic diversity probably includes reduced availability of high-flow refuge habitat due to limited bedform topography as bars are regularly graded.

Evaluation criteria for gravel bar grading in the Russian River are based on the relative amount of material removed and/or graded from the bar in the vertical dimension. The more material that is removed from bar above the thalweg, the greater the potential effects. Grading protocols that leave a greater percentage of the bar height intact will have smaller effects. By maintaining as much of the bar form as possible, alteration of channel morphology and hydraulics are reduced, including minimizing changes to width-depth ratio, sinuosity, roughness, sediment transport, velocity, and overall habitat complexity. This is consistent with the recent NOAA Fisheries guidelines for evaluating sediment removal (NOAA Fisheries 2003).

The proposed gravel bar grading procedures for the Russian River call for a post-graded bar that is at least 1.5 feet higher than the elevation of the low-flow channel, in order to maintain a thalweg. Additional procedures for maintaining a vegetated buffer strip adjacent to the low-flow channel and for grading the bar to prevent stranding are described in Section 4.4.

NOAA Fisheries (2003) guidelines suggest that an approach based on effective discharge to define the limits of gravel extraction from bars should be used in order to minimize harmful effects associated with gravel bar grading. This method is based on the concept that removal of bar material above the water surface elevation defined by the effective discharge (i.e., the flow that, over the long-term, transports the most sediment in the channel) represents the least potential harm. Bar material removed below the stage corresponding to the effective discharge represents a potentially greater harm to geomorphic and fish habitat conditions.

To address bank erosion problems, SCWA would conduct gravel bar grading activities in the Russian River in Sonoma County, and MCRRFCD would conduct these activities along a 36-mile reach of the Russian River. Defining the effective discharge at any given location, and the corresponding stage at a given bar cross-section site is not practical for evaluation purposes in this BA. Stage-discharge relationships are different at any particular site along the longitudinal profile of the river. The effective discharge also changes as the mainstem gains flows from major tributaries. Therefore, the evaluation criteria presented in this analysis is based on a more simplified approach that considers the relative percentage of the bar height that is likely to be removed (Table C-48).

Table C-48 Evaluation Criteria for Gravel Bar Grading in the Russian River

Category Score	Evaluation Criteria Category
5	No sediment removal or grading.
4	<25% of the bar height is removed.
3	25% to 50% of the bar height is removed.
2	>50% to 75% of the bar height is removed.
1	>75% of the bar height is removed.

As an example application of the criteria, assume that the maximum bar height is 6 feet above the adjacent thalweg and SCWA removes 4.5 feet of the bar height, leaving 1.5 feet above the thalweg. The score would be 2 ($4.5 \div 6.0 = 0.75$ or 75 percent).

In addition to the scoring criteria associated with bar height, a qualitative assessment is applied that considers the anticipated linear extent and frequency of maintenance actions, protocols implemented, and observations of existing aquatic habitat and geomorphic conditions.

The bar height scoring criteria are not applied to sediment maintenance activities in constructed flood control channels. Although there are similarities between sediment transport, channel hydraulics, and fluvial processes in the constructed flood control channels with natural channels, there are some important differences. A critical difference is that the width-depth ratio, entrenchment ratio, sinuosity, and sometimes gradient have been significantly altered from natural conditions in order to provide flood capacity. Therefore, the stage-discharge relationships associated with the "effective discharge" of an essentially permanently altered channel form is much different than the original natural channel form. Most of the geomorphic characteristics (dimension and planform) of the flood control channels are non-adjustable, and were purposely designed this way in order to ensure flood capacity.

C.1.8.5.3 Natural Channels

SCWA would, under certain site-specific catastrophic conditions, remove sediments in natural channels in the Russian River basin, including the Russian River. This sediment removal is usually done in conjunction with bank stabilization work. The range of activities associated with bank stabilization and sediment removal include levee repair, vegetation removal, and channel realignment where bars are directing high flows into unstable streambanks. Potential long-term habitat-altering effects of sediment and bank stabilization maintenance in natural channels include:

- Reduced canopy cover, increasing water temperatures.
- Reduced recruitment of spawning gravels.
- Change in channel geomorphology, including straightened channel platform that limits development of pool habitat, and overall simplification of habitat complexity.

Evaluation of habitat-altering effects of sediment maintenance activities in natural channels is based on a qualitative assessment that considers the extent and frequency of maintenance actions. Consideration is also given to the type of protections and BMP guidelines built into the SCWA approach for work in natural channels.

C.1.8.6 SEDIMENT MAINTENANCE

Sediments are removed from constructed flood control channels and are redistributed (i.e., bar grading) in natural channels with flood easements to ensure that channel capacity is maintained and to reduce bank erosion. Sediment maintenance takes place in constructed flood control channels. SCWA would not perform routine sediment removal activities in natural waterways. SCWA has hydraulic maintenance easements that are permissive and SCWA would continue to access various natural creeks to remove debris or vegetation to restore hydraulic capacity.

Habitat effects from sediment maintenance activities for listed species may include:

- Increased water temperatures and reduced cover if riparian vegetation is removed or disturbed;
- Reduced supply of spawning gravels;

- Change in channel geomorphology that may result in various habitat effects such as alteration of fish passage conditions, reduced channel sinuosity that limits pool and rearing habitat, and reduced high-flow refuge, and;

- General loss of hydraulic and associated aquatic habitat complexity depending upon the type of habitat conditions normally present in the project reach.

C.1.8.6.1 Flood Control Channels

Almost all sediment removal in Zone 1A[1]-constructed flood control channels is confined to streams draining to Laguna de Santa Rosa near Stony Point Road in the Rohnert Park-Cotati area. The combination of very flat gradients (typically less than 0.002 foot/foot) and high sediment production to the streams result in sediment deposition that reduces channel flood capacity.

Long-term changes to critical habitat for salmonids may occur due to sediment maintenance, but these effects could be either positive or negative. Long-term negative effects include potential reduction in available gravels that are of suitable size for spawning and lack of bed-form features such as bar-pool development that influence fish passage and also provide hydraulic diversity and associated habitat diversity. Long-term positive effects may include a reduction in fine sediment loading to downstream reaches that could improve spawning gravel quality, pool depth, and overall habitat diversity.

Evaluation of sediment removal effects in flood control channels is based on observations of changes in channel geomorphic and habitat conditions prior to and following excavation activities in 2000 and 2001. The status and recent history of sediment excavation for all Zone 1A-flood control channels, as well as ongoing hydraulic assessments, are used to estimate the extent of the work that is likely to be performed.

C.1.8.7 DEBRIS CLEARING

Debris clearing includes the removal of large woody debris, construction debris, and trash (e.g., shopping carts, tires, cars) from the stream channel to improve the flood capacity of the waterway. Equipment is operated from the bank rather than in the channel, so there is little risk for direct fish mortality attributable to debris-clearing activities. Changes to instream habitat may occur when debris is removed. large woody debris may be removed from flood control channels in Zone 1A under certain conditions, and is infrequently removed from natural channels only under catastrophic conditions.

Construction debris and trash may not always be harmful to salmonid habitat, and may even provide some of the only instream shelter available in a degraded urban stream. However, removal of trash or construction debris that can degrade water quality is beneficial. Furthermore, streams that are free of trash and filled with more natural instream cover elements are more aesthetically pleasing, and public clean-up events encourage stewardship of streams by local residents. Therefore, it is assumed that trash

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[1] For a description of SCWA Zones, see Section 1.4.7 of main draft report.

removal provides a net benefit, and only the effects of large woody debris removal are considered.

Large woody debris can play an important role in the structure and function of fish habitat, particularly in forested regions. In non-forested regions, large woody debris may be nonexistent, or have only a very limited scope of influence on channel and aquatic habitat conditions. Removal of large woody debris can potentially reduce the amount of instream cover, reduce pool frequency and depth, and simplify hydraulic and habitat diversity in the channel. Given the importance of woody debris to salmonid habitat, particularly large woody debris, evaluation criteria specifically address large woody debris removal.

The importance of large woody debris for salmonid habitat and biological productivity has been well-documented. Large woody debris provides cover and habitat diversity for salmonids and substrate for benthic invertebrates that serve as food (Sedell et al. 1984, 1988, Bisson et al. 1987). Large woody debris creates pools and undercut banks for cover, plays an important role in controlling stream channel morphology (Keller and Swanson 1979, Lisle 1986, Sullivan et al. 1987, cited in Hicks et al. 1991), and influences sediment movement, gravel retention, and composition of the biological community (Bisson et al. 1987 and Sullivan et al. 1987, cited in Murphy and Meehan 1991, Bilby and Ward 1989, cited in Flosi et al. 1998). Large woody debris creates hydraulic gradients that increase microhabitat complexity (Forward 1984) and the abundance of salmonids is often linked to the abundance of woody debris, especially in the winter (Bustard and Narver 1975, Tschaplinski and Hartman 1983, Murphy et al. 1986, Hartman and Brown 1987).

CDFG defines large woody debris as a piece of wood having a minimum diameter of 12 inches and a minimum length of 6 feet (Flosi et al. 1998). Root wads must be at least 12 inches in diameter at the base of the trunk but do not have to be six feet long.

Large woody debris may be found in the instream zone (stream channel within bankfull discharge demarcations), or in the recruitment zone. Although researchers have various means of identifying the width of the riparian zone from which large woody debris is recruited, in general terms it is no wider than the average height of the typical tree that borders the channel. Trees that are more distant from the channel than their average height, cannot be readily recruited into the channel as large woody debris. The recruitment zone represents approximately 70 percent of the large woody debris recruitment potential to the stream in natural forested channels (McDade et al. 1990, Forest Ecosystem Management 1993, cited in Flosi et al. 1998) so long-term management of the riparian zone is important. Evaluation criteria associated with vegetation control (Section C.1.8.3) address the extent of vegetation removal in the recruitment zone.

Evaluation criteria are structured to give high scores when large woody debris is not removed or if it is modified in place, but retained in the channel rather than completely removed. Modification, as used in this evaluation, includes cutting and removing a portion of the large woody debris. Alternatively, modification could include reorienting in the stream to prevent bank erosion or anchoring a piece of large woody debris so that it is not unstable. An intermediate score is given for limited removal practices (infrequent, only as necessary for flood control), and low scores when it is removed indiscriminately or entirely, and results in reduction of cover or other habitat functions provided by scour around large woody debris (Table C-49). Because the recruitment of large woody debris to a stream can be infrequent or episodic, even occasional removal of large woody debris has the potential to reduce the availability of high-quality salmonid habitat.

Table C-49 Large Woody Debris Removal

Category Score	Evaluation Criteria Category
5	No large woody debris removal or modification.
4	Large woody debris not removed, but modified.
3	Large woody debris removal limited to only when it poses a flood control hazard; removal does not result in substantial reduction of cover or scour in the area.
2	Large woody debris removal limited, but potentially results in moderate reduction of cover or scour.
1	Complete removal of large woody debris resulting in substantial reduction of cover or scour.

If large woody debris is removed in areas where spawning and rearing is likely to occur in either immediate or downstream areas, the effect would be greater than if large woody debris is removed from stream reaches that primarily function as a migration corridor. However, even in a migration corridor, large woody debris can provide cover and velocity breaks for migrating adults to rest and therefore should be retained as much as possible. In urban areas, the installation of instream structures may provide some of the benefits of large woody debris for migrating or rearing salmonids but still retain sufficient flood control capacity in the stream. Therefore, very infrequent removal of large woody debris may not result in a large reduction of cover or scour elements. As part of the scoring for activities associated with large woody debris removal, consideration is given to the primary habitat function of the channel where large woody debris is removed.

C.1.9 Criteria Related to Restoration and Conservation Actions

Restoration and conservation measures include actions to protect, restore, improve, or enhance aquatic habitat, riparian systems and stream and river channels, reduce the input of fine sediments to the stream corridors, and reduce water use. These actions could include preservation, restoration, or enhancement of riparian or aquatic habitat conditions and may have direct or indirect benefits to the coho salmon, steelhead, and Chinook salmon life-history stages in the Russian River system.

Direct actions are those that have an immediate response and utilization by a target species for one or more life-history stages. For example, actions that would have a direct benefit would include the addition of spawning gravel or the physical improvement of pool quality. The addition of spawning gravel would result in improved spawning conditions or increased spawning opportunities, and may increase recruitment to young-of-the-year (YOY) life-history stages. Improving pool quality could directly benefit several life-history stages for coho salmon and steelhead, including juvenile rearing, winter habitat conditions for juveniles, and summer low-flow refuge habitat. It could also improve adult holding habitat for all three species. Fish populations could show a rapid response to these actions because they involve direct improvements to the physical habitat in the stream.

An indirect action would require some time before it would be reflected in the fish population. For instance, implementing a sediment control plan in a tributary watershed may require many years before an improvement can be detected in the receiving waters, and it may be many more years before these improvements result in a response in the fish population. Reduction in fine sediment could affect habitat quality for invertebrates, alter the local temperature regime, and/or eventually improve spawning habitat quality. All these actions would result in indirect changes to habitat conditions supporting fish. Typically, indirect benefits are realized in incremental changes over time compared to direct benefits.

C.1.9.1 Evaluation Criteria for Restoration Projects

Planning and prioritization of restoration opportunities is an important component in an effective conservation program. Because financial resources are finite, it is important to maximize the biological benefit of conservation and restoration projects. This can be done by developing and utilizing information about the watershed, coordinating efforts on a basin-wide level, developing partnerships with other stakeholders, and seeking opportunities to bring additional financial or in-kind resources to the program.

An assessment of the biological value of specific SCWA restoration projects is made based on the size of the project, whether habitat or population data suggest the area is important or has the potential to be important for spawning or rearing, and the time-frame for the expected benefit. A qualitative assessment of the biological benefit score is given to each project based on several factors as outlined in Table C-50. Each project is evaluated for each of the target species, and for spawning/incubation, rearing, or migration life-history stages.

Typically, larger projects provide more benefits than smaller projects. The size limit is often constrained by funding sources, permitting issues, and the amount of work that can be accomplished in a single working season.

The conservation action is also qualitatively assessed based on the expected duration of the action and the time-frame to full development. The importance of short duration actions are considered to be lower than those with a 1-year to 3-year life span, and both are lower than actions that endure for more than 3 years. An additional time-frame to consider in evaluating the project is the time it takes the project to become fully functional at an ecological level. How long a project takes to become fully functional depends on the type of project. For example, the addition of spawning gravel to a system where spawning sites are limited can have an immediate and dramatic benefit. In contrast, it may take 10 to 20 years for the effects of a riparian restoration program to mature to the point where it begins to benefit the target species. Construction actions are typically evaluated without regard to full development time-frames. However, if riparian vegetation growth and development is required prior to full development, time-frames are typically on the order of decades. Projects with rapid start-up times may have a greater beneficial effect.

Table C-50 Components Considered in Determining the Biological Benefit of a Restoration Project

Component	Description
Size	Length of stream affected. Downstream or upstream habitat may also be affected. For example, streambank erosion control is likely to reduce sedimentation of downstream habitat, or installation of fish passage may result in access to miles of upstream spawning and rearing habitat.
Time	The time-frame for expected benefits. Projects with rapid start-up times may have a greater beneficial effect. Some projects may take some time before benefits are fully realized, but if they are of long duration or permanent in nature, substantial benefits can be realized for protected fish species.
Habitat Elements	A qualitative assessment of the habitat elements affected and their relative importance to protected fish species.
Habitat or Population Data	If data are available on population abundance or stream assessments, they are used to assess the relative importance of the project to protected fish species. For example, a fish passage project that provides access to several miles of high-quality spawning and rearing habitat may have more value than instream habitat improvements in an area that is likely to have limited rearing or spawning habitat. If a known limiting factor is addressed, the project is considered to have a higher benefit.
Cost	Limited public and private funds are available for restoration actions. Projects that can deliver the most benefit for these dollars are preferred alternatives.
Education	A project that has an educational component or can serve as a demonstration project may have indirect beneficial effects.

A qualitative assessment is made of project effects on critical habitat, including elements such as canopy cover, instream cover, sediment effects, and bank erosion. Because changes in these elements are difficult to quantify, actual scoring criteria have not been developed for them. If the project addresses a known limiting factor, it would be considered more important. Any stream assessments or population data that have been collected are used to evaluate the current or potential use of the project area by salmonids.

A project that has an educational component or can serve as a demonstration project may have indirect beneficial effects. Because so much of the Russian River watershed is privately owned, landowner cooperation is essential. Demonstration projects serve to educate the public about the advantages of a restoration action and may help alleviate concerns related to work involving endangered or threatened species on private property.

A restoration project may improve the quantity and/or quality of the habitat or it may produce no change. If a conservation action is expected to improve habitat for protected species, it is scored 3 or better (Table C-51). A project that has a very high potential to benefit would be one that improves a large portion of valuable or potentially valuable habitat, preferably with effects that are likely to occur soon and for an extended period of time. Intermediate or small-sized projects may also have high potential to benefit, and are assigned a score of 4. Some projects, while useful, have small, localized benefits but may be located in areas that have less value for salmonid spawning or rearing and are assigned a score of 3. While a particular project may not have a direct benefit within its footprint, it may provide upstream benefits (such as access to spawning habitat) or downstream benefits (such as an improvement in water quality). If a project has little or no benefit but uses limited public financial resources that may be better spent elsewhere, it is assigned a score of 2. A project that is poorly planned or implemented, results in long-term degradation of habitat, or wastes limited financial resources is scored a 1. For example, a large streambank stabilization project that places riprap along a streambank or redistributes gravel bars to protect a landowner's property, but that degrades salmonid habitat, would receive a lower score than a bank stabilization project designed to improve riparian and instream habitat for salmonids.

Table C-51 Biological Benefit Evaluation Criteria for Restoration Actions

Category Score	Evaluation Criteria Category
5	Very high potential to benefit.
4	High potential to benefit.
3	Moderate potential to benefit.
2	No benefit and uses scarce resources.
1	Poorly planned or implemented, degrades habitat.

C.1.9.2 WATERSHED MANAGEMENT PROJECTS

Watershed management projects include three general categories of projects: 1) data collection, 2) demonstration projects, and 3) information coordination and dissemination. Studies that collect data on salmonids and their habitat have indirect effects that may not be quantifiable, but are potentially significant. When conservation activities are coordinated with other agencies, a greater benefit may accrue to protected species and their habitat. Demonstration projects gather and utilize information on methods to meet specific goals and make that information available for application on a wider scale than the immediate project area, possibly on a watershed scale. Public information and public involvement activities may also have unquantifiable but important effects on listed species and their habitat.

C.1.9.3 DATA COLLECTION

In the absence of data, it would be very difficult to protect listed species and their habitat. Studies funded, coordinated, or implemented by SCWA produce information that is essential for effective and cost-efficient restoration and conservation activities.

C.1.9.3.1 Habitat Data

Information that can potentially benefit critical habitat include stream habitat surveys, water quality data, and temperature data. Stream habitat surveys have been conducted throughout the watershed to help identify and prioritize streams that are in need of restoration. Results are summarized in the CDFG 2002 Draft Russian River Basin Fisheries Restoration Plan (CDFG 2002). Water temperature monitoring since 1996 will help identify streams that may provide the best thermal conditions for salmonid rearing. Finally, a baseline reference for water quality has been established in selected streams, and this will be used to determine relative water quality status in other streams.

C.1.9.3.2 Fish Population Data

Insufficient data exist to assess salmonid population trends in the Russian River. A comprehensive population monitoring program developed in conjunction with CDFG and NOAA Fisheries will assess the current status of steelhead and coho salmon, particularly juvenile abundance. Furthermore, the Inflatable Dam/Wohler Pool fish sampling program is producing information on smolt emigration in the late spring. Of particular value has been information about Chinook salmon.

Historically, coho salmon, steelhead, and Chinook salmon from distant watersheds have been planted in the Russian River. Hatchery broodstock, has, until recently, included out-of-basin stocks. Potential genetic effects of out-of-basin transfers are outlined in a benefit/risk analysis developed for the DCFH and CVFF (FishPro and ENTRIX, Inc. 2000). It is probable that many tributaries of the Russian River contain a mixture of native and non-native stocks. Genetic information is being evaluated to determine which streams may contain relatively native genetic strains so that those strains can be targeted for conservation.

C.1.9.3.3 Invasive Plant Species

Non-native plants can alter the riverine ecosystem. When invasive plant species replace native riparian vegetation, alterations can occur to the food web, riparian and instream cover, and general habitat characteristics of the stream. One non-native species on the Russian River is the aforementioned *Arundo donax*.

Arundo is potentially a serious problem for the Russian River ecosystem. It has already established itself in extensive portions of wetland and riparian habitats, especially in southern California. *Arundo* forms tall, dense stands. It propagates vegetatively more so than by seed, and it has deep roots. It quickly invades new areas, particularly where the ground is cleared, when floods break up clumps of *Arundo* and transport rhizomes downstream. Because it outcompetes native species, it may jeopardize riparian restoration projects.

Arundo changes the stream channel by retaining sediments and constricting flow. Root masses that can be more than a meter thick stabilize streambanks and alter flow regimes. *Arundo* provides less canopy cover than native species do, so stream temperatures are increased (Dunne and Leopold 1978, cited in Bell 1997). It changes the quality and timing of organic debris that forms the base of the riparian food chain. It does not seem to provide food or habitat for native species of wildlife, including salmonids. It is highly flammable, and when riparian corridors are changed from flood-defined to fire-defined communities, diverse ecosystems are converted to pure stands of *Arundo* (Bell 1997).

Because *Arundo* has not yet established itself to the devastating degree that it has in southern California, a proactive removal program may still be effective and affordable in the Russian River watershed. It is effectively removed with a combination of manual or mechanical means and herbicide use. Additional treatments may be needed to prevent it from re-establishing itself. As *Arundo* spreads in a downstream direction, eradication has to be coordinated in the watershed. Furthermore, a public information campaign is required, as *Arundo* is sold in nurseries.

While some information about *Arundo* and its control has been developed in southern California, insufficient data exist for northern California streams. Its biology and ecology is not well-studied. The mechanisms with which it overtakes native riparian communities are not well-understood, particularly in cooler northern climates. It is not known what factors may prevent infestation. Distribution and abundance data are lacking. More information is needed to develop effective *Arundo* eradication and prevention programs.

C.1.9.4 Demonstration Projects

Demonstration projects provide information that can be applied throughout the Russian River watershed to improve conditions for salmonids and their habitat. They include Pierce's Disease control, fish-friendly farming, and the Palmer Road Erosion control.

C.1.9.4.1 Pierce's Disease Control

Pierce's Disease is caused by a bacterium (*Xylella fastidiosa*) that kills grapevines. It is spread by xylem-feeding insects in the sharpshooter family, particularly the glassy-winged sharpshooter. There is no known control for Pierce's Disease. Management focuses on control of the sharpshooter and removal of diseased plants. The most susceptible vines are on the outskirts of grape-growing areas next to pastures or riparian areas.

In the Russian River watershed, vineyards are often located adjacent to riparian zones where the vegetation is prime habitat for sharpshooters. When vineyard owners indiscriminately clear riparian vegetation, valuable riparian corridors can be destroyed. By removing only host plants that attract sharpshooters and leaving others, the insects' abundance can be dramatically reduced. Plants that sharpshooters favor include wild grape, Himalayan blackberry, French broom, and periwinkle. Plants that are not likely to attract sharpshooters include oaks, California bay laurel, alder, maple, ash, and red willows (University of California at Davis 1999).

C.1.9.4.2 Fish-Friendly Farming

Many streams in the Russian River watershed run through agricultural land, particularly vineyards. The success of this voluntary education and certification program depends on the level of participation and implementation by growers. Therefore, incentives that increase participation are important to the success of the program.

C.1.9.4.3 Palmer Road Erosion Control

Roads can cause degradation of streams by modifying natural drainage and accelerating erosion processes, altering channel morphology and by changing the runoff characteristics of watersheds. (Furniss et al. 1991). The resulting sedimentation of streams can be dramatic. Improperly designed roads can affect migration of salmonids. There are guidelines for road siting, building, and maintenance that can help reduce negative effects and minimize sedimentation of streams (Furniss et al. 1991, WDFW 1999, NMFS 2000). A properly designed rural road can provide a demonstration of principles that should be applied throughout the watershed.

C.1.9.5 Evaluation Criteria for Information Value

Some research data may have localized usefulness, such as water quality sampling conducted in specific streams. Other research may be useful to many areas of the river watershed, such as development of effective *Arundo* eradication methods. Evaluation criteria for information gathering or dissemination assess how wide a geographic area the information has the potential to be used in, and a qualitative assessment is made on the relative biological benefit to listed species or designated critical habitat (Table C-52).

Table C-52 Information Value Evaluation Criteria

Category Score	Evaluation Criteria Category
5	Basin-wide applicability.
4	A region or "type" of habitat (i.e., small tributaries or lower mainstem).
3	Isolated project/stream information.
2	Information not useful to protected species or habitat.
1	Incorrect or misleading information.

C.1.9.6 Information Coordination and Dissemination

The Russian River basin will be subjected to increasing demands on its resources. If it is to be protected and restored to the fullest extent possible, coordination among stakeholders is essential. By coordinating with agencies, government entities, and various organizations or watershed groups, limited resources can be put to maximum use. By providing information and training to the public, additional conservation actions can be implemented and future problems can be avoided. Furthermore, some activities, such as *Arundo* control or development and implementation of water quality standards, must be coordinated on a watershed level to be fully effective. While the benefits of these activities are not quantifiable, they are potentially significant.

C.1.10 Fish Production Facility Criteria

The ESA's focus is on natural populations and the ecosystems upon which they depend. However, the ESA recognizes that artificial propagation may be useful for recovery efforts. NOAA Fisheries has noted that (Hard et al. 1992):

> *Artificial propagation of Pacific salmon may be consistent with the purposes of the Endangered Species Act in two situations: 1) when artificial propagation facilitates the recovery of a listed species, or 2) when the enhancement of unlisted populations does not impede the recovery of a listed species or compromise the viability or distinctiveness (and hence be a factor in the listing) of an unlisted species.*

Artificially spawned populations (that may or may not be listed) may affect naturally-spawned populations. NOAA Fisheries notes that deliberations over the use of artificial propagation for recovery must recognize the potential for deleterious direct and indirect effects on the listed species (Hard et al. 1992).

This section identifies potential effects hatcheries may have on listed fish species and their habitat. There are two basic categories of potential effects: 1) effects on water quality, and 2) genetic and ecological effects on protected fish populations. Evaluation criteria are presented that reflect the range of hatchery practices that can affect these factors. It must be emphasized that the identified potential effects are intended to describe hatcheries in general and not the specific operations at the Don Clausen Fish Hatchery (DCFH) and Coyote Valley Fish Facility (CVFF). The evaluation criteria are developed to describe a range of hatchery operating procedures, and as such will describe operations in addition to those currently practiced at the DCFH and CVFF.

C.1.10.1 Water Quality

Operations at most hatchery facilities involve diversion of water into the facility and discharge back to the river. In concentrated fish production processes, waste solids from fish feces and excess feed typically become entrained in the hatchery water supply system. If not treated, the effluent from fish production facilities can affect water quality in the receiving water (the stream) through increased turbidity, settleable solids, biological oxygen demand (BOD), and nutrient loading. Additionally, water temperature in the discharged water can potentially affect salmonid habitat.

Aquatic animal production facilities with more than 20,000 pounds annual production are subject to discharge water quality limits established through the EPA National Pollution Discharge Elimination System (NPDES). For the Russian River area, NPDES permits are administered by the NCRWQCB. The NCRWQCB has established water quality limits for the areas it administers based on designated beneficial uses for the subject waters. In Dry Creek and the Russian River, these beneficial uses include cold-water fish life, which reflects the general water quality standards for the protection of threatened species.

NPDES permits require that the facilities be equipped with waste treatment equipment to insure compliance with specified water quality criteria (Table C-53). Compliance is monitored by sampling the facility effluent two times per month, with results submitted in a monthly report to the NCRWQCB. It is further stipulated that sampling occur during cleaning operations, because this is when poor water quality conditions are most likely to occur.

Table C-53 Discharge Standards for DCFH and CVFF

Parameter	Effluent Limit (Daily Maximum)
Total Suspended Solids	15 mg/l
Total Settleable Solids	0.2 ml/l/hr
pH	within 0.5 of receiving waters
Salinity (chloride)	250 mg/l
Temperature	no measurable change to receiving water
Turbidity	no increase > 20% of background
DO	> 7.0 mg/l
Flow – DCFH	15.5 million gallons/day
Flow – CVFF	7.11 million gallons/day

The discharge permits include stipulations in addition to the monthly monitoring noted above. For example, discharge of wastes from pond cleaning and the bypass of wastes around the pollution control pond are prohibited. At DCFH, it is prohibited to discharge detectable levels of chemicals used for the treatment or control of disease, other than salt (sodium chloride).

Because the NPDES discharge standards reflect general water quality requirements for coho salmon, steelhead, and Chinook salmon, they provide a practical means for assessing potential effects from DCFH and CVFF operations. Evaluation criteria for water quality effects are presented in Table C-54. A score of 5 is given to facilities that are in full compliance with NPDES standards and lower scores are given when routine compliance occurs less frequently.

Discharge standards have not been set for some chemicals used in captive breeding programs. The primary chemicals used are vaccines, antibiotics, chlorine, along with chlorine neutralizer, and salt to address disease concerns. Sex hormones are used to increase development, and anesthesia is typically used whenever captive animals are handled.

Table C-54 Water Quality Compliance Evaluation Criteria

Category Score	Evaluation Criteria Category
5	Continuous compliance with NPDES standards.
4	Compliance with 75 – 99% of standards.
3	Compliance with 50 – 74% of standards.
2	Compliance with 25 – 49% of standards.
1	Compliance with 0 – 24% of standards.

The waste stream of a broodstock program is disinfected once a parasite or contagion is confirmed. Effluent streams flowing from infected stocks can have chlorine gas injected into the outfall flowing from the facility. The chlorine, after adequate time, is then neutralized with sodium thiosulfate. If chlorine injection fails, the waste stream should be shut down.

The DCFH coho salmon captive broodstock program qualifies for NOAA permit exemption for the use of chemotherapeutics through qualification under the Threatened and Endangered Species permit process. However, this applies only to the use of vaccines and treatments on listed salmon and does not exempt the operator from state or federal environmental compliance. Vaccines, antibiotics, disinfectants, and hormones must be removed from the effluent stream. Carbon filtration is the conventional method used.

C.1.10.2 GENETIC AND ECOLOGICAL RISKS

The potential risks of hatchery production on protected fish species can be categorized into two areas: genetic risks and ecological risks. Genetic risks include loss of diversity within and between populations, outbreeding depression, and inbreeding depression. Ecological risks include increased competition for food, habitat or mates; increased predation; disease transfer; altered migration behavior; decreased long-term viability; artificial selection; disproportional survival; and harvest bycatch. Each of these risk factors is described in this section, along with a discussion of the major theoretical and observed effects to wild salmonid populations. There is also discussion of the general hatchery practices and management decisions that may affect each risk factor. Table C-55 is a cross-reference index that associates the various risk factors with hatchery practices.

Stock productivity refers to the mean number of offspring produced per adult that survives to spawn. A stock productivity less than 1.0 means the population is decreasing; greater than 1.0 means it is increasing; and equal to 1.0 means it is stable. For large populations, the stock productivity can be less than 1.0 (declining) for a relatively long time before the population goes extinct. Small populations have less time. Salmon populations are characterized by extreme variation in year-to-year returns, and it is not uncommon for healthy populations occasionally to experience stock productivity less than 1.0 (Flagg et al. 2000). Conversely, a severely depressed population may occasionally experience a year with productively greater than 1.0, only to return to the depressed state the following year.

When stock productivity is consistently less than 1.0, a supplementation program may be implemented as an interim measure to maintain the population until environmental conditions change or anthropogenic effects are corrected. While supplementation may have some negative effects on wild fish, these are preferable to extinction.

Table C-55 Risks to Wild Salmonids from Hatchery Production and Associated Operations That May Contribute to Each Risk

Risks to Wild Salmonids Associated with Hatchery Production	Hatchery Operations That May Contribute to Each Risk[1]						
	Source of Broodstock	Numbers of Broodstock Collected	Broodstock Sampling and Mating	Rearing Techniques	Release Strategies	Duration in Hatchery Captivity	Harvest Management
Genetic Risks							
Loss of Diversity							
Within Population Diversity	X	X	X	X		X	
Between Population Diversity	X				X		
Outbreeding Depression	X		X				
Inbreeding Depression	X	X					
Ecological Risks							
Competition					X		
Predation					X		
Disease Transfer				X			
Outmigration Behavior					X		
Long-Term Viability				X			
Artificial Selection	X			X		X	
Disproportional Survival		X	X				
Harvest Bycatch							X

[1] See next section, E.1.10.3, for discussion.

Supplementation may not be needed for a depressed stock with productivity greater than 1.0, as it should be able to recover on its own. A supplementation program may increase the population size above some critical level more rapidly and therefore accelerate recovery.

The analysis of the hatchery program is conducted in large part following the structure of Waples (1996). First, an overview of genetic and ecological risks is provided. This is followed by a description of evaluation criteria used to assess the level of risk associated with specific hatchery practices.

C.1.10.3 Genetic Risks

C.1.10.3.1 Loss of Diversity

Conservation biologists divide genetic and life-history diversity into two components, diversity within a population and diversity between populations. Within-population diversity is the suite of phenotypes within an interbreeding group of individuals. Between-population diversity is that component of variation that occurs between populations. Both types of variation are widely viewed as crucial contributors to long-term species persistence. Maintenance of within-population diversity is crucial for the continued existence of a given population because it provides the ability to respond to random environmental changes (stochasticity) typically encountered in the local environment. Maintenance of between-population diversity allows the species to inhabit environments with different selective regimes. It is crucial for evolutionary processes that help ensure the long-term survival of the species.

Loss of Within-Population Diversity

Loss of within-population diversity refers to a loss of genetic variability within the composite (hatchery-reared and naturally-spawned) population. Genetic drift (random changes in gene frequencies over time) and inbreeding (which may be a result of small population size within either population component) may reduce within-population genetic variation. Hatchery programs can contribute to the loss of within-population diversity when hatchery broodstock includes only a limited sample of the gene pool available in the naturally spawning population, and when the survival of that component is magnified.

Changes in within-population diversity are primarily influenced by four factors: the selection of broodstock sources; the numbers of broodstock available to both the hatchery-reared and naturally spawning population components; the degree to which the sampling methods represent those spawning aggregates; and the survival of adults and/or juveniles during their period of residency in the hatchery. Specific recommendations for minimizing the potential risk associated with each of these functional areas of hatchery operations are provided.

Loss of Between-Population Diversity

Loss of between-population diversity refers to a reduction in the genetic differences between discrete populations. It should be noted that a reduction in between-population diversity does not necessarily mean there is a reduction of variation within any population (although this may be the case); it merely describes the loss of genetic identity (loss of divergence) between two or more populations. In salmon, the primary natural mechanism leading to a loss in between-population diversity is straying and successful reproduction of fish in non-natal streams.

Anadromous salmonids exhibit a high degree of fidelity to their natal stream when returning as adults to spawn (Quinn et al. 1985). Although approximately 90 percent of spawning adults return to their natal stream, some stray to non-natal streams to reproduce (Grant 1997). Straying is thought to facilitate colonization of new habitat (Unwin and Quinn 1993), maintain genetic diversity within small populations (Grant 1997), and help maintain single populations at risk of extinction (Hill et al. 2002). There is a considerable genetic influence on survival and return of salmon, local fish can have a survival advantage over translocated fish, and population-specific adaptation can occur fairly rapidly (within 30 generations) of establishment of populations into uncolonized habitat (Unwin et al. 2003).

High rates of straying may lead to genetic homogenization, or a decrease in genetic differences between populations (Grant 1997). Straying may reduce the total amount of genetic variability within salmon stocks if it leads to a loss of alleles through genetic drift. Alternately, straying could have a positive influence if it increases the range of heritable phenotypic expression within populations. Whatever the case, management may seek to maintain between-population variability by minimizing straying of hatchery-reared individuals to nontarget populations. Release strategies that may reduce the incidence of straying are discussed in Section C.1.10.5.4.

Stock transfers, the active collection of fish from one population for use as broodstock or for direct outplanting in another population, directly erodes between-population diversity. Rather than maintaining a natural stray rate, stock transfers may constitute a substantial portion of the adult returns in the next generation. If stock transfers are repeated, the target stock may become genetically similar to the source population. As discussed previously, this may not be inherently detrimental. However, adaptations to the local environment, such as age at return or other life-history traits, may be disrupted by too much straying or by stock transfers (Quinn 1984). It is theorized that several populations, each with unique genetic characteristics, may have a greater combined resiliency to environmental change than several genetically identical populations (Cooper and Mangel 1998).

Given that it may be impossible to protect between-population diversity within the Russian River basin, managers should seek to avoid erosion of this variation on a larger geographic scale. To do so, managers should seek to maintain the highest possible degree of homing fidelity for hatchery raised fish released in the Russian River. Straying can be minimized by rearing and acclimating hatchery fish in water from the target stream (Dittman et al. 1994, 1996). Thus, the development of acclimation facilities could reduce the risk of loss of between-population diversity.

C.1.10.3.2 Outbreeding Depression

Outbreeding depression refers to a decrease in fitness following hybridization of individuals with divergent genetic composition. Outbreeding depression may occur due to dysgenic processes (i.e., the breakdown of coadapted gene complexes [epistasis]) or through disruption of genotype-environment interactions (i.e., loss of local adaptation). The source of hatchery broodstock is the primary factor affecting the risk of outbreeding depression. A breeding program that derives broodstock for a target population from its returning adults is unlikely to cause outbreeding depression. However, if programs utilize stock transfers, the risk of outbreeding depression increases as geographic distance and differences in selective regimes between stocks increase.

Intrinsic Coadaptation (Epistasis)

Intrinsic coadaptation, or epistasis, describes traits that rely on interactions between genes/loci (Lynch 1991). Templeton (1986a, 1986b) indicates that coadaptation occurs most readily in species with restricted recombination, a possible result of population subdivision, small population size, and inbreeding. Since salmonids exhibit a strong homing extinct (i.e., return to the natal stream), it is conceivable that populations may develop unique coadaptations resulting in a fitness advantage in the local environment. It follows that introgression by individuals not possessing the same unique coadaptation could disrupt the epistatic interaction and decrease fitness of the progeny. Unfortunately, there are no tests currently available to easily assay the existence or probability of the existence of coadaptations.

The transmission of coadaptations from parent to progeny makes direct measures of coadaptation difficult as well. For example, F_1 (first generation) progeny arising from a cross between an individual possessing a coadaptation and one lacking a coadaptation will inherit the epistatic interaction. However, recombination during gamete formation will likely disrupt the coadaptation, and it will not be passed to the F_2 (second generation) progeny. Therefore, breakdown of epistasis typically will not occur until the F_2 generation, and, to date, few studies have overcome the difficulties of tracking fish through two full generations. Gharret and Smoker (1991) documented a decrease in fitness exhibited by F_2 crosses of even- and odd-year pink salmon. The authors suggest that the decrease in fitness may have resulted from the breakdown of a coadapted gene complex. Unfortunately, the authors did not incorporate sufficient F_2 controls, so their assertion that decreased fitness was the result of disruption of epistasis remains unproven.

Certainly, the life-history characteristics of salmonids suggest that the evolution of population-specific epistatic interactions is possible. However, the inability to assess the existence or assign a probability of occurrence to coadaptation limits management implications to a qualitative discussion. Overall, the probability of outbreeding depression increases with reproductive isolation between the stocks in question. For example, if broodstock is derived from adult returns to the target population, outbreeding depression is almost impossible. However, if broodstock is derived as a stock transfer from a distant population with which natural gene flow is currently and was historically minimal, the probability of outbreeding depression increases.

Extrinsic Coadaptation

In addition to breakdown of coadapted gene complexes, outbreeding depression can occur as a consequence of hybridization between populations that express different karyotypes. Karyotype refers to the number of chromosomes possessed by an individual, while karyotypic race refers to the distribution of karyotypes within a population.

Successful hybridization of salmon with different karyotypes is documented (Kusunoki et al. 1994). For example, Thorgaard (1983) found that coastal stocks of rainbow trout that were indistinguishable morphologically or by allelic frequency, varied in chromosome number from 58 to 64 within and between putative populations. However, while hybridization of karyotypic races occurs, Garcia-Vazquez et al. (1995) suggest that wild fish undergo selection toward a standard karyotype. While outbreeding depression doesn't always occur as a result of hybridization of salmonids with differing karyotypes, management may seek to avoid mixing different karyotypic races of salmonids. In order to avoid mixing karyotypic races, hatchery programs could derive broodstock from the endemic population. Whatever the case, since individuals with differing karyotypes may occur within the same population, it is unclear whether or not outbreeding depression will occur as a result of hybridization between fish with differing karyotypes.

Outbreeding Depression due to Disruption of Local Adaptation

The mechanisms of outbreeding depression discussed previously are dysgenic (strictly genetic) in nature. Outbreeding depression may also occur via disruption of local adaptation. Local adaptation refers to a phenotype (either physical or behavioral) resulting from the complex interaction between a genotype and the environment. An illustration of this type of outbreeding depression is provided by coho salmon hatchery practices in coastal Oregon streams. Broodstock collection proceeded by the capture of fish as they appeared in the river, and continued until broodstock quotas were achieved. The result was selection for the earliest returning adults. Since run-timing is a partially heritable trait, hatchery-reared progeny returned and spawned earlier than the mean return time of the stock prior to hatchery influence (Nickelson et al. 1986). Early spring freshets may have reduced the survival of progeny of early returning fish relative to those returning at the historical peak (Nickelson et al. 1986). Avoiding outbreeding depression as a result of loss or disruption of local adaptation could be achieved by deriving broodstock as a representative sample of the target population.

Because salmonids exhibit a strong homing instinct and exhibit age-structured or overlapping generations, it is possible that populations may develop unique coadaptations, karyotypes, or local adaptations. It follows that a population possessing a unique coadaptation, karyotype, or local adaptation may be negatively affected by introgression from a genetically divergent stock. Therefore, the best mitigation for outbreeding depression is the derivation of broodstock as a representative sample of the target population. Theoretically, supportive breeding following this practice would prohibit mixture of genetically divergent individuals, minimizing the risk of outbreeding depression.

Since genetic divergence requires reproductive isolation or strong differential selection, geographically proximate stocks experiencing similar selective regimes may be more genetically similar than two stocks that are geographically distant and/or subject to different selective pressures. In addition, gene flow in the form of straying is more likely between proximate stocks, which suggests that proximate stocks may be less genetically divergent than geographically distant stocks.

The risk of outbreeding depression as it relates to hatchery operations can be affected by the source of broodstock and by the protocols for broodstock sampling and mating. The anticipated risk of these operations with respect to the proposed Russian River production program alternatives are discussed in Section A1.10.5.

C.1.10.3.3 Inbreeding Depression

Inbreeding refers to the mating of two closely related individuals; more closely related than any two individuals collected randomly from the population (nonrandom mating). Inbred progeny have a lower heterozygosity (i.e., two alleles at any single locus are more likely to be the same than expected under random mating) than the population as a whole (Tave 1993). For example, progeny of sib-mating (spawning of a brother-sister pair) suffer a mean loss of 25 percent heterozygosity relative to heterozygosity expected from a randomly mating pair (Waldman and McKinnon 1993). Inbreeding results in a reduction in the overall heterozygosity of a population. Inbreeding is not inherently positive or negative, and is frequently used in aquaculture to increase productivity or optimize traits (Shields 1993, Tave 1993, Wangila and Dick 1996).

Inbreeding depression occurs when decreased heterozygosity lowers fitness through loss of heterozygote advantage (heterosis), loss of adaptively advantageous alleles, or by expression of alleles that are deleterious in the homozygous state (dominance) (Allendorf and Leary 1986, Mitton 1993, Lutz 1996 and 1997, Ballou 1997, and David 1997). The effects of inbreeding may vary depending on the demographic history of a population. For example, populations that have experienced serial bottlenecks may not be as susceptible to inbreeding, since deleterious alleles may have been purged during the bottleneck events (Tanaka 1997) and/or there is very little genetic variation left to lose. Recent evidence suggests that inbreeding depression may be exacerbated by fluctuating or stressful environments (Miller 1994). Thus, it is conceivable that environmental perturbations and fluctuating population sizes within the Russian River might promote inbreeding depression.

Within the scientific community, there is little agreement as to the probability or extent of inbreeding depression among fishes. For example, according to Waldman and McKinnon (1993), inbreeding depression has been detected in every fish species for which there are data. However, most of their examples refer to intensive aquaculture programs for which inbreeding may be intentional or the result of poor hatchery management. Therefore, this represents a test of the theory of inbreeding depression, not a test of the likelihood of occurrence in a natural population or a well-managed conservation program. Among salmonids, it has been suggested that residual tetraploid inheritance could limit the potential for deleterious effects of inbreeding. Because of residual tetraploidy, the expression of a deleterious recessive allele would require that an individual be homozygous at all four loci (if the second pair of chromosomes are active) (Waples 1990), or, alternatively, that selectively advantageous alleles are lost at each locus. Regardless, inbreeding in general should be avoided in conservation programs to the extent possible, unless specifically required as a means to maintain unique variation (e.g., half-sib mating strategies).

In the Russian River, the risk of inbreeding depression can be managed to some extent by the selection of broodstock. Populations used for broodstock acquisition could be sampled for genetic diversity and the degree of relatedness expressed within potential founding populations could be used as a means to prioritize sources. Once broodstock is obtained, pairings could be assigned based (in part) on minimum kinship methods (Montgomery et al. 1997). Therefore, the relative degree of risk posed by inbreeding depression can be evaluated based on the implementation of broodstock collection protocols that maximize diversity and employment of minimum kinship methods to formulate spawning matrices.

C.1.10.4 Ecological Risks

C.1.10.4.1 Competition

Competition between hatchery-reared and naturally-spawned population components may occur if resources are limited (e.g., food, shelter, etc.). If the number of hatchery-reared juveniles released into a stream exceeds its carrying capacity, competition may negatively affect naturally-spawned fish. Competition may occur between individuals of the same species (intraspecific competition) or between individuals of different species (interspecific competition). Intraspecific competition in salmonids may be greater because there is greater niche overlap between conspecifics (members of a species) than individuals of different species. However, management attempts to increase the population size of any one of the three listed species may affect the others.

Smolt size and survival are positively related, and this has encouraged hatchery practices that release large smolts. However, the naturally spawning component of the population is likely to have evolved under the ecological conditions (temperature and food) present in a particular watershed. Release of larger hatchery fish may change the competitive pressures on naturally-spawned fish, as well as change the long-term selective pressure by the mere presence of larger conspecifics in the ecosystem over time.

Data to adequately quantify competitive effects for Russian River salmon are lacking. Therefore, evaluation criteria focus on release strategies. Risk aversion techniques provided through release strategies used at other hatchery facilities to reduce competition are discussed.

C.1.10.4.2 Predation

The release of hatchery-reared juveniles can affect production among the naturally spawning population component through direct predation. For example, studies have shown that large hatchery-reared juveniles may consume smaller naturally reared juveniles of the same (Sholes and Hallock 1979) or different species (Cannamela 1992, 1993). Flagg and Nash (1999) suggest only releasing hatchery-reared juveniles that are a similar size to naturally-spawned juveniles as a means of limiting direct predation. If supportive breeding increases the number of adult returns, predation of juveniles of the same or other species may likewise increase. Predation may be an unavoidable consequence of increased population size.

Release strategies may also play an important role in limiting predation. For example, traditional release strategies typically involve the direct release of a large number of hatchery-reared fish in one location at one time. This strategy may result in the attraction or concentration of predators at the site of release. Furthermore, hatchery releases are typically performed during daylight hours, which may make juveniles more vulnerable to visual predators (Flagg and Nash 1999).

Implementing a production goal that produces only smolts and no fingerlings, and further allows the smolts to acclimate and volitionally leave the facility, may result in decreased predation. Fish leaving an acclimation site volitionally may be physiologically prepared for emigration to saltwater, minimizing residence time in the freshwater environment (Pascual et al. 1995). This has the dual benefit of minimizing predation of the hatchery-reared fish by freshwater predators, and minimizing predation of freshwater fish by hatchery-reared juveniles (Flagg and Nash 1999). There is also evidence that territorial aggressiveness decreases with the onset of smoltification (Iwata 1996).

Evaluation criteria for predation are based on the risk-aversion techniques provided through release strategies used at other hatchery facilities to reduce predation.

C.1.10.4.3 Disease Transfer

Biological pathogens, which are the causative agent of a disease, are an integral part of the environment of all animals, including both wild and cultured fish populations. A NOAA Fisheries review of the ecological impacts of hatcheries (Flagg et al. 2000), indicates that almost all pathogenic microorganisms existed in wild fish populations before the introduction of hatcheries. Since the hatchery environment has a greater potential for higher rearing densities and stress than the natural environment, hatchery populations may be a reservoir for infectious agents. However, there is little evidence to suggest that disease transmission to wild stocks is either routine or significant.

Infectious agents may also be transmitted from fish to other fish through intermediate hosts. Examples of this process include the oligocheate worm, *Tubifex tubifex*, which is responsible for whirling disease, and birds, which can spread pathogens through their feces after ingestion of infected fish (Flagg et al. 2000).

Policies have been developed to prevent the spread of pathogens that might result in the release of seriously infected salmon from hatcheries. These policies are discussed in the rearing techniques discussion of the evaluation criteria section.

C.1.10.4.4 Outmigration Behavior

The "pied piper effect" refers to the downstream schooling of wild fish influenced by large numbers of downstream migrant hatchery fish. It has been reported by scuba divers, but there is little documentation regarding the frequency or condition of its occurrence. There are no quantitative studies of the effects of such behavior on survival of Pacific salmon species or on differential survival between hatchery and wild fish (Flagg et al. 2000). Studies of hatchery-reared Atlantic salmon recorded a smolt-to-adult survival (SAR) of 6.8 percent when fish were released to streams during peaks of smolt outmigration, whereas the SAR was only 2.6 percent for fish released during the troughs, suggesting a possible benefit of the schooling phenomenon (Flagg et al. 2000).

C.1.10.4.5 Long-Term Viability

There are concerns over the long-term viability of reintroduced or hatchery supplemented stocks (Meffe 1992). As noted previously, the interaction of wild-spawned and hatchery-reared individuals is unpredictable. However, experimental evidence suggests that post-release survival of hatchery-reared fish may be low. Reisenbichler and McIntyre (1977) suggest that survival of juvenile steelhead to emigration was higher among wild-spawned fish than their hatchery-reared conspecifics. In addition, Skaala et al. (1996) found that hybrid (hatchery-reared x wild) brown trout smolts had significantly lower survival than wild smolts. The authors employed a visual marker that confounded the results, as it may have increased predation of the hybrids. However, despite a 4:1 hatchery-reared to wild ratio, the genetic contribution of spawning hatchery-reared individuals was only 19.2 and 16.3 percent in two streams (Skaala et al. 1996). It should be noted, however, that parents of the hatchery-reared individuals were collected in a mountain lake and may therefore have lacked some adaptation to the riverine environment. Lane et al. (1990) found that emigration survival of hatchery-reared pink salmon smolts was only 0.4 percent compared to 1.4 percent for smolts of wild origin. Currens et al. (1997) and Williams et al. (1997) found that rainbow trout introduced to the Deschutes and Metolius rivers were derived from a coastal stock that lacked resistance to infection by *Ceratomyxa shasta* in contrast to the native stock. The authors indicate that the lack of resistance resulted in exclusion of introduced fish from areas occupied by the parasite. However, this is a contentious study, as there is some question as to the actual degree of resistance exhibited by the native and introduced stocks. Leider et al. (1990) found that introduced steelhead suffered lower survival probabilities at all lifestages than did the wild stock. Finally, Bachman (1984) found that the behavior of hatchery-reared brown trout was less efficient than wild conspecifics.

Examples of successful introductions or supplementation programs suggest that hatchery-reared fish may be successful under natural conditions. For example, Quinn et al. (1998) and Kinnison et al. (1998) reported on the results of a single introduction of Chinook salmon to New Zealand. The transplanted stock has successfully radiated into resident and migratory life histories, with possible divergence in run-timing among the anadromous stock. In addition, Clifford et al. (1998) found that farm-raised Atlantic salmon were successful in completing their life-cycle and breeding with wild conspecifics.

Unfortunately, many studies assessing the performance of hatchery-reared fish lack the data necessary to determine the factors resulting in success or failure of supplementation or reintroduction. In fact, most available data are from studies of traditional rather than supplementation hatchery programs. In addition, data are lacking to determine if wild-spawned progeny of hatchery-reared parents suffer a competitive disadvantage. These data are crucial for objective assessment of the long-term viability of supplemented populations.

When possible, supplementation programs should seek to obtain broodstock locally to avoid outbreeding depression and to take advantage of regional adaptations that may exist. In addition, it should be clear that supplementation programs cannot succeed in creating naturally sustaining populations unless the factor(s) responsible for the initial population decline are addressed. Documented examples of supplementation programs that addressed both of these principles were not found. Research is currently underway in the Pacific Northwest, but it will take time before these questions are answered.

C.1.10.4.6 Artificial Selection

Artificial selection refers to changes in genetic, behavioral, or phenotypic attributes resulting from rearing in an unnatural environment. As it pertains to hatcheries, artificial selection refers to selection for or against specific phenotypes (the expression of genetic variation as mediated by the environment) of broodstock and/or their progeny during residence in the hatchery. Domestication refers to the end-product of extreme artificial selection, typically requiring several generations, after which the cultured organisms have been altered genetically and behaviorally to the point that they are optimally suited only for the hatchery environment. Both processes result from either relaxation of natural selection in the hatchery environment and/or selection for phenotypes that are adaptive in the hatchery. For example, for a female of a given size, there is a tradeoff between the number and size of eggs that can be produced for a given energy allotment. Under natural conditions, fewer but larger eggs may be preferable, since the increased yolk reserve may provide a survival advantage to the progeny. Alternatively, in the hatchery, egg size may be irrelevant (within reason), since the eggs and juveniles are reared in a less stressful environment. Under such conditions, the relaxation of artificial selection in the hatchery could increase the prevalence of females with smaller, but more numerous eggs, since they would produce a relatively larger number of offspring, who themselves may exhibit decreased egg size. If this trait were to become a dominant feature among hatchery origin females, it is possible that hatchery-reared females would become less successful at reproducing in the wild relative to naturally-spawned females that produce larger eggs. Such a result is termed “domestication.” In general, artificial selection is viewed as a negative side effect of hatcheries, since selection (either deliberate or inadvertent) for phenotypes of greater value in the hatchery environment may decrease the prevalence of phenotypes that are more useful under natural conditions.

Artificial selection that is deliberate, such as selection for larger females for use as broodstock, is largely avoidable. However, artificial selection can be incidental. For example, culling eggs or juveniles exhibiting a high titer for bacterial kidney disease may result in inadvertent selection against those individuals possessing a natural resistance to the disease. In such a situation, practicing artificial selection may be preferable, since the alternative might be a massive horizontal transmission of a deadly disease. In addition, artificial selection can be directional or dispersive; that is, hatchery-rearing may consistently select for a median phenotype, or stochastically select throughout the range of a given reaction norm.

In general, the risks associated with artificial selection are governed by the magnitude of the selective gradient between the hatchery environment and the stream, the heritability of traits subject to artificial selection, and the period of residence in the artificial environment. All things being equal, if the magnitude of the selective differential between the hatchery and natural environment is large and directional, phenotypic divergence between hatchery-reared and naturally-spawned fish may occur rapidly. For example, run-timing among hatchery-reared fish can be dramatically and rapidly altered by selecting broodstock from the earliest adult returns. This occurs because managers can exert strong selection for early return by excluding later returning adults. However, the selection differential can be expected to result in lasting phenotypic divergence only if there is a heritable component of genetic variation governing the expression of the trait. Measures of heritability vary greatly by trait. Age of spawning is heritable to a degree of approximately 90 percent (Tave 1993), while smolt survival is approximately 1 percent heritable (Withler et al. 1987). For salmonids, the mean heritability of 264 traits listed by Tave (1993) is 27 percent. Therefore, for an average trait, approximately 73 percent of the expression (or phenotype) is governed by the environment. In terms of hatchery management, this means that even with 100 percent selection for a given trait (e.g., run-timing), one can expect approximately a 27 percent change in the location of the mean phenotype (e.g., a 27 percent shift toward earlier run-timing) per generation. All things being equal, one would expect the number of diverged traits, and the magnitude of divergence to increase with the duration of captivity. Simply stated, with a longer period of duration, more life-history stages may be subjected to artificial selection, and more traits may become susceptible to the effects of artificial selection. Finally, the accumulation of phenotypic divergence is related to the duration of artificial selection. That is, multiple artificial selection events will almost certainly have a greater effect than single events. For example, for programs that derive broodstock from hatchery-reared adult returns, the risk of serious accumulation of artificially selected phenotypes is greater than for programs that derive broodstock from naturally-spawned adult returns.

As it relates to artificial production, the risks of artificial selection can be minimized (though likely not completely avoided) by: 1) decreasing the selective gradient between the hatchery and natural environment; 2) minimizing the residence time of adults and or progeny in the hatchery environment; and 3) minimizing repeated artificial selection. These strategies are discussed in greater detail in the evaluation criteria sections relating to rearing techniques, duration in hatchery captivity, and source of broodstock, respectively.

C.1.10.4.7 Disproportional Survival

It has been suggested that increased survival of hatchery-reared fish results in a disproportional representation of the genomes of hatchery-reared parents (Leary et al. 1993). It is further theorized that disproportional representation decreases the effective population size of the hatchery-reared and target population (Leary et al. 1993). It is likely true that parental genomes incorporated in hatchery programs are represented disproportionately in the target stocks because the goal of supplementation programs is to increase egg-to-smolt survival. However, if the gene pool of the hatchery-reared component is indistinguishable from the naturally reared component, disproportional representation will have no detrimental effect. Whatever the case, decreases in effective population size due to disproportional survival of hatchery-reared fish may occur in two ways: unrepresentative broodstock collection, or disproportionate family contribution.

Hatchery-reared fish enjoy a survival advantage from egg-to-smolt transformation compared to wild-spawned conspecifics. Therefore, the progeny of hatchery-reared fish will likely contribute more adult returns in subsequent generations than progeny of a wild stock of similar size (assuming that the smolt-to-adult return is equivalent between groups). If the hatchery-reared component of a supplemented population is not genetically and behaviorally representative of the target population (unrepresentative broodstock collection), allelic frequencies may shift, or changes in life-history traits may occur. Therefore, it is crucial that broodstock collection is conducted such that the genetic, physical, and behavioral characteristics of the wild population are represented among the broodstock. This is a difficult task, however, there are examples of success. For example, the captive broodstock for the winter-run Chinook salmon program at Bodega Bay Marine Lab was representative of the wild stock, and supplementation has had positive effects on the effective population size of the wild population component (Arkush et al. 1997). If a population is large, random selection of broodstock across the adult return may be sufficient to ensure a representative broodstock. However, as population sizes decrease, the probability of selecting related individuals increases, and methods to determine descent (such as pedigree analysis) may become necessary.

A second factor contributing to decreased effective population size among hatchery-reared and naturally-spawned components is variance in family size. Variance in family size can occur through several mechanisms. For example, it is well-established that egg fertility varies among individual females. It follows that females with higher fertility may contribute more adults in subsequent generations if smolt-to-adult return is constant among progeny of all females. However, recent research indicates that even in naturally reproducing populations, variability in reproductive success is inherently extreme and only partially genetically based. For example, stochastic environmental events may favor individuals spawning at certain times in one year and at different times in subsequent years. The result is that in natural systems, a few individuals may give rise to a disproportionate number of offspring to represent the next generation (Laikre et al. 1998, Li and Hedgecock 1998). If hatcheries derive a representative broodstock yearly from the target population, the resulting increase in fitness of the hatchery-reared individuals may not be an added variance component beyond what would be observed by natural spawning. Further, if a hatchery broodstock is representative of the wild population it will be supplementing, the effects of variable reproductive success can be partially mitigated by equalizing the number of progeny released from each family (Allendorf 1993). Particularly in small populations, this practice would only reduce the effective population size if all naturally spawning individuals experienced similar levels of reproductive success. Overall, with proper management, increased survival among the hatchery component of a supplemented population should not be deleterious.

C.1.10.4.8 Harvest Bycatch

Overexploitation of naturally-spawned adults may occur when fisheries are targeted toward more abundant hatchery-reared adults. Obviously, where harvest is prohibited, the probability of overexploitation is minimal; however, when harvest is condoned, overexploitation may occur through direct harvest, or incidental mortality. If harvest is permitted, the probability of overexploitation is greater if quotas are based on the absolute number of returning adults rather than the number of naturally-spawned returning adults. While overexploitation may still occur, visually marking hatchery-reared progeny allows managers to target hatchery-reared adults in the fishery by allowing retention of only marked fish.

C.1.10.5 Evaluation Criteria for Genetic and Ecological Risks

In the previous sections, genetic and ecological risks of hatchery production were discussed with respect to the major theoretical and observed effects to wild salmonid populations. Table C-55 provided a cross-reference index that associates the risk issues to the various hatchery practices and management decisions that have the potential to affect those risks. This section provides the analysis approach that is used to assess these risks. Due to the diversity of hatchery operations, this discussion is organized into seven categories that encompass functional requirements. Following each hatchery operations category is a table that ranks the risk of various hatchery practices. The evaluation will focus on the process of implementing the most effective management approach for achieving recovery of listed fish species.

C.1.10.5.1 Sources of Broodstock

The source(s) of broodstock for the Russian River should be guided, in part, by the following considerations:

- Can salmonids captured within the Russian River and tributaries be expected to provide the genetic and life-history diversity necessary to successfully repopulate the Russian River and eventually provide for self-sustaining natural populations?

- Can restoration efforts be aided by importation of steelhead from nearby basins, or is the scale of local adaptation smaller (e.g., between spawning aggregates within a basin)?

- Do alternate broodstock sources inhabit watersheds with selective regimes similar to the Russian River?

An isolated harvest program would derive all broodstock from the supply of adult salmonids returning to the hatchery. For supplementation and integrated harvest programs, the annual source of broodstock would come from the wild population, wherever possible. The selection of a broodstock source for a supplementation or integrated harvest program ultimately will be dictated by availability. Within the constraints of availability, the following priorities are recommended:

1. Naturally-spawned broodstock collected in the most unbiased manner possible from the local target population(s), provided that collection of broodstock does not endanger the population.

2. Naturally-spawned adults from the nearest watershed, provided that collection of broodstock does not endanger the population. If several such sources are available, managers may wish to choose the location(s) that have a high probability of maintaining transfers, and that most closely match the environmental characteristics of the Russian River and tributaries. Further, when possible, managers may wish to consider using cryopreserved milt from local sources, if and when available, to fertilize the eggs of transferred females.

3. Hatchery-reared adults collected in the most unbiased manner possible from the local population.

4. Hatchery-reared adults from the nearest watershed, provided that broodstock collection is not unduly prohibitive for the donor hatchery program. If several such sources are available, managers may wish to choose the location(s) that have a high probability of maintaining transfers, and whose natural habitat most closely matches the environmental characteristics of the Russian River and tributaries. Further, when possible, managers may wish to consider using cryopreserved milt from local sources, if and when available, to fertilize the eggs of transferred females.

5. Naturally-spawned broodstock collected from within the same Evolutionarily Significant Unit (ESU), following the criteria described in priority two, above.

6. Hatchery-reared adults collected from within the same ESU, following the criteria described in criteria three, above.

7. Naturally-spawned broodstock or juveniles collected from a different ESU, following the criteria described in priority two, above.

8. Hatchery-reared adults or juveniles collected from a different ESU, following the criteria described in criteria three, above.

In general, the list of priorities is ordered from the least-to-most risky, and highest-to-lowest probability of success. This determination is based on the fact that the scale, magnitude, and biological significance of local adaptation are not fully understood. It follows that adults derived from the target population(s) would be the most ideal candidates for supplementation or integrated harvest programs, since these individuals may exhibit phenotypes that have been influenced by the natural selection regime of the local environment (i.e., locally adapted). Therefore, it is expected that the probability of short-term supplementation or integrated harvest success may be lower if adults are transferred from other watersheds, and the degree to which success may be hindered could be proportional to the distance of the transfer (Reisenbichler 1988). However, if the local population(s) is severely depressed, it is possible that the genetic and life-history variation present in the stock may be insufficient for adaptation to current and changing conditions within the Russian River watershed (Newman and Pilson 1997). For example, in small populations subject to high rates of random mortality, genetic drift may exert a stronger influence than natural selection (Adkison 1995). In such cases, it is possible that the potential benefits of using the local population as a source (e.g., potential local adaptation) of adults may be offset by low overall genetic variation, and that a transfer of exogenous genetic variation may provide a broader genetic background upon which natural selection can act (see Lewontin and Birch 1966), although this is not always the case (Leberg 1993).

The potential to use an existing hatchery stock as a donor further complicates the list of broodstock priorities. While the use of an existing hatchery stock has been at least partially successful in some cases (Phillips et al. 2000), this strategy should be approached cautiously (except for the isolated harvest alternative). The history of stock transfers and the management of the donor hatchery program should be reviewed in detail to determine if adults exhibit the ability to reproduce and function in the natural environment. For example, several studies have demonstrated that hatchery-reared fish may exhibit decreased survival and productivity relative to natural-origin fish under natural conditions (e.g., Fleming and Gross 1993), although research has also demonstrated that this is not always the case (Rhodes and Quinn 1999). Therefore, the decision to use a hatchery as a source of adults or juveniles versus naturally-spawned adults or juveniles from a more distant location should be carefully considered.

The source of broodstock used in hatchery operations has the potential to affect the wild population primarily through the mechanism of outbreeding depression. Depending on specific circumstances, the source of broodstock also has potential to contribute to loss of within-population or between-population diversity, inbreeding depression, or straying of the hatchery-reared component. However, by utilizing local stocks as the source of broodstock, the source of genetic material in the first-generation hatchery component is presumably identical to that of the wild population.

In cases where the abundance of local stocks are insufficient to meet the broodstock demand, then the priorities noted at the beginning of this section can provide the basis for selecting alternative sources while minimizing risk to the wild population. Table C-56 organizes the recommended priorities for broodstock source into five categories and provides each category with a score of relative risk level. Since the local stock is the recommended source of broodstock for both supplementation and integrated harvest production alternatives, there is no difference in risk level between these alternatives and the no production alternative. However, until such time that there are adequate numbers of wild salmonids to assure that broodstock harvest would not affect the target stock, it is recommended that a mix of hatchery-reared and naturally spawning broodstock be utilized to satisfy the minimum broodstock goals.

Table C-56 Source of Broodstock Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	Local broodstock source (target stock), collected in the most unbiased manner possible.
4	Naturally-spawned broodstock source from the nearest watershed; or a combination of naturally-spawned and hatchery-reared broodstock from the local source.
3	Hatchery-reared broodstock source from the local or nearest watershed; or naturally-spawned broodstock source from within the same ESU.
2	Hatchery-reared broodstock source from within the same ESU; or naturally-spawned broodstock source from a different ESU.
1	Hatchery-reared broodstock source from a different ESU.

C.1.10.5.2 Numbers of Broodstock

Escapement Goals Based on Genetic Criteria

Escapement and broodstock goals are based on probabilities associated with maintaining genetic variation and limiting demographic risks, both to the hatchery-reared and naturally-spawned components of the Russian River population(s). Except for an isolated harvest alternative, the escapement and broodstock goals are formulated to provide for "genetic and life-history redundancy;" that is, in the event that a brood year is lost either in the hatchery as a result of catastrophic failure, or in the stream as a result of a random environmental event, the surviving component should maintain sufficient genetic and life-history variation to maintain the stock. To reasonably ensure this redundancy requires that the hatchery and naturally spawning population components are representative of one another both genetically and in life-history characteristics, and that both components are maintained at a large size, which may be unrealistic in the short-term for a listed species. In addition, it is implicitly assumed that hatchery practices will not grossly alter life-history characteristics. This assumption may or may not hold true as discussed elsewhere in this document.

Within the body of peer-reviewed conservation literature there exists a wide range of population sizes deemed acceptable for the maintenance of genetic and life-history diversity. These estimates range from as low as 500 adults per generation (or approximately 167 adults per year in the case of coho salmon, Soulé 1980) to as high as 1,000 spawning adults per year (Rieman and Allendorf 2001) for naturally spawning populations. With regards to hatchery programs, as few as 200 adults can provide the basis for a reasonably healthy broodstock (Allendorf and Ryman 1987).

Most estimates of minimum viable population size are based on maintenance of genetic variation. Although the maintenance of genetic variation is certainly not the only consideration for conservation programs, it is widely accepted that the maintenance of genetic variation is critical for maintaining fitness and the ability to respond to a changing environment. It is also widely accepted that the maintenance of genetic variation is linked to population size (Frankham 1995a, 1995b), although this relationship is sometimes obscure (Kelly 2001).

To formulate escapement goals for Russian River steelhead and coho salmon, an estimate was made of the number of adults per year required to maintain a 95 percent probability that alleles occurring at a frequency of 1 percent or greater would be retained for three generations (15 years) for steelhead, five generations (15 years) for coho salmon, and three generations (15 years) for Chinook salmon within both the hatchery and natural components of the Russian River watershed. This rate of retention is supported as a reasonable goal for conservation programs (Allendorf and Ryman 1987, Kapuscinski 1991). A period of 15 years was selected to allow the completion of the habitat restoration activities that will likely be required to improve sustainability of naturally spawning Russian River salmonids. Rates of allele retention are based on the following binomial equation:

$$P_R = 1-(1-p)^{2(\text{generation length})(Nb)}$$

Where:

P_R = probability of maintaining a rare allele

p = frequency of the rare allele

N_b = yearly effective population size

Over a period of several generations the probability is described as:

$$P_{RC} = P_R^{\,G}$$

Where:

P_{RC} = cumulative probability of maintaining a rare allele

G = number of generations

To utilize these equations, an estimate of the ratio of effective number of breeders (N_b) to the census population size N in a given year is required. Estimates of both of these parameters were not available specifically for Russian River salmonids, so estimates from the literature were employed. Within the literature, effective population size (N_e) to census population size (N) ratios averaged 0.10 across taxa (Frankham 1995b), and varied from 0.04 to 0.83 among salmonids (Simon et al. 1986, Bartley et al. 1992). Average effective population sizes for salmonids are reported to be 0.2 (Allendorf et al. 1997), 0.3 (McElhany et al. 2000), or 0.33 (Hedrick and Hedgecock 1994). In the following sections, estimates are provided of the population size necessary to maintain a 95 percent probability of retaining alleles at a frequency of 1 percent or greater, over a period of one to five generations, assuming a range of N_e/N ratios. To provide a conservative estimate, final escapement goals are based on an N_e/N ratio of 0.2. Finally, to provide estimates of yearly broodstock and escapement needs, N_e/N ratios were transformed to N_b/N ratios by dividing N_e by the mean generation length of steelhead (5 years), and coho salmon and Chinook salmon (3 years) following Waples (1990).

It should be noted that several simplifying assumptions are necessary to utilize this methodology. Obviously, it must be assumed that the range of effective population size ratios is appropriate for Russian River salmonids. Second, the precision of allele retention estimates spanning more than one generation is greatly effected by the stability of allele frequencies. If allele frequencies fluctuate substantially between generations, the precision of allele frequency estimates will be low. Nonetheless, these methods will likely provide a reasonable estimate for managers until such time as Russian River-specific data are available. Third, strictly speaking, these estimates are applicable primarily to neutral genetic variation; however, additive genetic variation for quantitative traits might be expected to follow a similar pattern, since it is linked to heterozygosity, and is expected to decline proportionally (Falconer and Mackay 1996). Fourth, this method assumes that all individuals in a population are equally likely to be sampled. In practice, this assumption may not hold true, particularly if there is a high risk that a few families will be over-represented by inherent biases in sampling protocols and/or difficulties associated with randomly sampling small, geographically dispersed populations.

Finally, these estimates should be regarded as minimum escapement estimates. In particular, escapement goals based on genetic criteria may be far too small to provide an ample buffer against random environmental stochasticity experienced by naturally spawning fish.

Instream and Hatchery Broodstock Escapement Goals

As mentioned previously, escapement goals are formulated to provide genetic redundancy. To achieve this goal, instream spawning and hatchery broodstock goals are formulated to provide a 95 percent probability of retaining alleles occurring at a frequency of 1 percent or greater within each component of the Russian River population for a period of three steelhead generations (15 years), five coho salmon generations (15 years), and five Chinook salmon generations (15 years). Such a strategy assures (probabilistically) that genetic variation will be maintained even if one of the two population components suffers complete reproductive failure. Assuming an N_b/N ratio of 0.2, approximately 200 adult steelhead spawners (Table C-57), approximately 400 adult coho salmon spawners (Table C-59), and approximately 230 adult Chinook salmon spawners (Table C-60) are required in each environment to meet this goal. Tables C-58, C-60 and C-62 list the number of adult steelhead, coho salmon, and Chinook salmon spawners, respectively, necessary to maintain genetic variation assuming various N_b/N ratios. If coho salmon juveniles are collected to form a captive brood, juvenile collections would have to be substantially larger to provide 420 adults for broodstock. Assuming a 40 percent fry to adult survival rate[2] in captivity (Arkush et al. 1997), approximately 588 fry would be required to achieve the adult spawning goal within the hatchery.

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[2] Juvenile collection thus far has focused on more advanced life-history stages. Fry survival estimates were used to provide a conservative estimate.

Table C-57 Number of Steelhead Adults Required for Broodstock to Maintain a 95 Percent Probability of Maintaining Alleles at a Frequency of 1 Percent or Greater for 15 Years (3 Steelhead Generations) (Assuming an N_b/N Ratio of 0.2)

Broodstock Size	Nb	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
50	10	0.63	0.40	0.25	0.16	0.10
100	20	0.87	0.75	0.65	0.56	0.49
150	30	0.95	0.90	0.86	0.82	0.78
200	40	0.98	0.96	0.95	0.93	0.91
250	50	0.99	0.99	0.98	0.97	0.97
300	60	1.00	1.00	0.99	0.99	0.99
350	70	1.00	1.00	1.00	1.00	1.00
400	80	1.00	1.00	1.00	1.00	1.00
450	90	1.00	1.00	1.00	1.00	1.00

Table C-58 Estimated Rates of Allele Retention (at a Frequency of 1 Percent or Greater) for Broodstocks of 100, 200, and 400 Steelhead Adults (Assuming a Range of N_b/N Ratios)

Nb/N	Nb	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
Broodstock of 100 Adults						
0.1	10	0.634	0.402	0.255	0.162	0.102
0.2	20	0.866	0.750	0.650	0.562	0.487
0.3	30	0.951	0.904	0.860	0.818	0.778
0.4	40	0.982	0.964	0.947	0.930	0.913
0.5	50	0.993	0.987	0.980	0.974	0.968
0.6	60	0.998	0.995	0.993	0.990	0.988
0.7	70	0.999	0.998	0.997	0.996	0.996
Broodstock of 200 Adults						
0.1	20	0.866	0.750	0.650	0.562	0.487
0.2	40	0.982	0.964	0.947	0.930	0.913
0.3	60	0.998	0.995	0.993	0.990	0.988
0.4	80	1.000	0.999	0.999	0.999	0.998
0.5	100	1.000	1.000	1.000	1.000	1.000
0.6	120	1.000	1.000	1.000	1.000	1.000
0.7	140	1.000	1.000	1.000	1.000	1.000

Table C-58 Estimated Rates of Allele Retention (at a Frequency of 1 Percent or Greater) for Broodstocks of 100, 200, and 400 Steelhead Adults (Assuming a Range of N_b/N Ratios) (Continued)

Broodstock of 400 Adults						
0.1	40	0.982	0.964	0.947	0.930	0.913
0.2	80	1.000	0.999	0.999	0.999	0.998
0.3	120	1.000	1.000	1.000	1.000	1.000
0.4	160	1.000	1.000	1.000	1.000	1.000
0.5	200	1.000	1.000	1.000	1.000	1.000
0.6	240	1.000	1.000	1.000	1.000	1.000
0.7	280	1.000	1.000	1.000	1.000	1.000

Table C-59 Number of Coho Salmon Adults Required for Broodstock to Maintain a 95 Percent Probability of Maintaining Alleles at a Frequency of 1 Percent or Greater for 15 Years (Five Coho Salmon Generations) (Assuming an N_b/N Ratio of 0.2)

Broodstock Size	N_b	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
50	10	0.45	0.21	0.09	0.04	0.02
100	20	0.70	0.49	0.34	0.24	0.17
150	30	0.84	0.70	0.58	0.49	0.41
200	40	0.91	0.83	0.75	0.69	0.63
250	50	0.95	0.90	0.86	0.82	0.78
300	60	0.97	0.95	0.92	0.90	0.87
340	68	0.98	0.97	0.95	0.94	0.92
350	70	0.99	0.97	0.96	0.94	0.93
400	80	0.99	0.98	0.98	0.97	0.96
450	90	1.00	0.99	0.99	0.98	0.98

Table C-60 Estimated Rates of Rare Allele Retention (Occurring at a Frequency of 1 Percent or Greater) for Broodstocks of 100, 200, and 400 Coho Salmon Adults (Assuming a Range of N_b/N Ratios)

Broodstock of 100 Adults						
N_b/N	**N_b**	**One**	**Two**	**Three**	**Four**	**Five**
0.1	10	0.453	0.205	0.093	0.042	0.019
0.2	20	0.701	0.491	0.344	0.241	0.169
0.3	30	0.836	0.699	0.585	0.489	0.409
0.4	40	0.910	0.829	0.754	0.687	0.625
0.5	50	0.951	0.904	0.860	0.818	0.778
0.6	60	0.973	0.947	0.922	0.897	0.873
0.7	70	0.985	0.971	0.957	0.943	0.929
Broodstock of 200 Adults						
Nb/N	**Nb**	**One**	**Two**	**Three**	**Four**	**Five**
0.1	20	0.701	0.491	0.344	0.241	0.169
0.2	40	0.910	0.829	0.754	0.687	0.625
0.3	60	0.973	0.947	0.922	0.897	0.873
0.4	80	0.992	0.984	0.976	0.968	0.960
0.5	100	0.998	0.995	0.993	0.990	0.988
0.6	120	0.999	0.999	0.998	0.997	0.996
0.7	140	1.000	1.000	0.999	0.999	0.999
Broodstock of 400 Adults						
Nb/N	**Nb**	**One**	**Two**	**Three**	**Four**	**Five**
0.1	40	0.910	0.829	0.754	0.687	0.625
0.2	80	0.992	0.984	0.976	0.968	0.960
0.3	120	0.999	0.999	0.998	0.997	0.996
0.4	160	1.000	1.000	1.000	1.000	1.000
0.5	200	1.000	1.000	1.000	1.000	1.000
0.6	240	1.000	1.000	1.000	1.000	1.000
0.7	280	1.000	1.000	1.000	1.000	1.000

Table C-61 Number of Chinook Salmon Adults Required for Broodstock to Maintain a 95 Percent Probability of Maintaining Alleles at a Frequency of 1 Percent or Greater for 15 Years (5 Generations) (Assuming an N_b/N Ratio of 0.2)

Broodstock Size	Nb	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
50	10	0.63	0.40	0.25	0.16	0.10
100	20	0.87	0.75	0.65	0.56	0.49
150	30	0.95	0.90	0.86	0.82	0.78
200	40	0.98	0.96	0.95	0.93	0.91
230	46	0.99	0.98	0.97	0.96	0.95
250	50	0.99	0.99	0.98	0.97	0.97
300	60	1.00	1.00	0.99	0.99	0.99
350	70	1.00	1.00	1.00	1.00	1.00
400	80	1.00	1.00	1.00	1.00	1.00
450	90	1.00	1.00	1.00	1.00	1.00

Table C-62 Estimated Rates of Allele Retention (at a Frequency of 1 Percent or Greater) for Broodstocks of 100, 200, and 400 Chinook Salmon Adults (Assuming a Range of N_b/N Ratios)

Nb/N	Nb	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
Broodstock of 100 Adults						
0.1	10	0.634	0.402	0.255	0.162	0.102
0.2	20	0.866	0.750	0.650	0.562	0.487
0.3	30	0.951	0.904	0.860	0.818	0.778
0.4	40	0.982	0.964	0.947	0.930	0.913
0.5	50	0.993	0.987	0.980	0.974	0.968
0.6	60	0.998	0.995	0.993	0.990	0.988
0.7	70	0.999	0.998	0.997	0.996	0.996
Broodstock of 200 Adults						
0.1	20	0.866	0.750	0.650	0.562	0.487
0.2	40	0.982	0.964	0.947	0.930	0.913
0.3	60	0.998	0.995	0.993	0.990	0.988
0.4	80	1.000	0.999	0.999	0.999	0.998
0.5	100	1.000	1.000	1.000	1.000	1.000
0.6	120	1.000	1.000	1.000	1.000	1.000
0.7	140	1.000	1.000	1.000	1.000	1.000

Table C-62 Estimated Rates of Allele Retention (at a Frequency of 1 Percent or Greater) for Broodstocks of 100, 200, and 400 Chinook Salmon Adults (Assuming a Range of N_b/N Ratios) (Continued)

Nb/N	Nb	Generation One	Generation Two	Generation Three	Generation Four	Generation Five
Broodstock of 400 Adults						
0.1	40	0.982	0.964	0.947	0.930	0.913
0.2	80	1.000	0.999	0.999	0.999	0.998
0.3	120	1.000	1.000	1.000	1.000	1.000
0.4	160	1.000	1.000	1.000	1.000	1.000
0.5	200	1.000	1.000	1.000	1.000	1.000
0.6	240	1.000	1.000	1.000	1.000	1.000
0.7	280	1.000	1.000	1.000	1.000	1.000

Minimum Escapement Goal

Calculating a minimum escapement estimate is not as straightforward as adding escapement goals for each population component. In a sense, escapement of 420 steelhead adults (210 for natural spawning and 210 for broodstock) or 840 coho salmon adults (420 for natural spawning or 420 for broodstock) would be sufficient to meet instream spawning and broodstock needs for a given year. However, the number of adults contributing to the next generation is expected to be higher for adults spawned in the hatchery than adults spawning in the natural environment. Therefore, given that natural production is likely below replacement for naturally spawning salmonids in the Russian River, substantially more adults will be required for natural spawning to ensure that escapement is sufficient in the next generation.

Further, it is assumed that hatchery rearing will provide a substantial survival benefit relative to natural spawning. As a result, escapement in the next generation is expected to be much higher for hatchery-reared adults relative to naturally-spawned adults if the population components are of roughly equivalent size in the present generation. Therefore, escapement for natural spawning may have to be much higher than the minimum estimates derived above, to provide for the infusion of naturally-spawned broodstock into the hatchery program. In addition, far more hatchery-reared adults will be available for natural spawning and hatchery rearing.

Ryman et al. (1994) provide a simple equation to account for differential reproductive success experienced by adults in the hatchery versus natural environment. The variance-effective population size (N_e)[1] of the combined population (including adults of hatchery

[1]Variance-effective population size is one component of total effective population size N_e. Strictly speaking, all estimates of N_e discussed in this plan refer to variance-effective population size.

and natural origin) expected to return in the next generation, can be estimated by the following equation:

$$N_b=(1-1/2N)/(1/N'^2(N'+(N'_c(N'_c-0.5)/N_c)+(N'_w(N'_w-0.5)/N_w))-1/N)$$

Where:

N_b = total yearly effective population size at time t+1

N_c = N_b of the broodstock for the hatchery-reared component

N'_c = N_b returning in the next generation as a result of hatchery rearing

N_w = N_b of the naturally spawning population component

N'_w = N_b returning in the next generation as a result of natural spawning

N = total population size at time t

N' = total population size in the next generation (t+1)

For example, using the escapements formulated previously for steelhead, (210 spawners in each environment), and assuming an adult return rate of 0.5 and 4, respectively, for instream and hatchery spawners, contribution to the next generation would be 105 and 840 adults of natural and hatchery origin, respectively. Ultimately, for reasons relating to artificial selection, it would be appropriate to maintain a minimum escapement goal that would allow for collection of broodstock solely from naturally-spawned adults. With a natural return rate of 0.5, this would require instream spawning by a minimum of 420 adults to return 210 adults to the next generation (which could be a combination of natural and hatchery origin individuals). Therefore, assuming 5 percent prespawn mortality in each environment, a minimum escapement of 630 adults per year would be required for a steelhead supplementation or integrated harvest program.

Using the escapements formulated previously for coho salmon (420 spawners in each environment), and assuming an adult return rate of 0.5 and 4, respectively, for instream and hatchery spawners, contribution to the next generation would be 210 and 1,680 adults of natural and hatchery origin, respectively, with a combined N_b of 168 as determined by the equation above. Ultimately, for reasons relating to artificial selection, it would be appropriate to maintain a minimum escapement goal that would allow for collection of broodstock solely from naturally-spawned adults. With a natural return rate of 0.5, this would require instream spawning by a minimum of 840 adults (which could be a combination of natural and hatchery origin individuals). Therefore, assuming 5 percent prespawn mortality in each environment, a minimum escapement of 1,322 adults per year is required.

Using the escapements formulated previously for Chinook salmon (242 spawners in each environment), and assuming an adult return rate of 0.5 and 4, respectively, for instream and hatchery spawners, contribution to the next generation would be 121 and 968 adults of natural and hatchery origin, respectively. Ultimately, for reasons relating to artificial selection, it would be appropriate to maintain a minimum escapement goal that would allow for collection of broodstock solely from naturally-spawned adults. With a natural return rate of 0.5, this would require instream spawning by a minimum of 484 adults to return 242 adults to the next generation (which could be a combination of natural and hatchery origin individuals). Therefore, assuming 5 percent prespawn mortality in each environment, a minimum escapement of 726 adults per year would be required for a Chinook salmon supplementation or integrated harvest program.

The numbers of broodstock used in hatchery operations has the potential to affect the wild population primarily through the mechanisms of inbreeding depression and loss of within-population diversity. However, by determining and using the minimum number of broodstock necessary to maintain the genetic variability of the population, the risk of genetic effect is minimized. Table C-63 organizes the potential range broodstock availability into five categories with a score of relative risk level.

Table C-63 Numbers of Broodstock Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	Maintenance of N_b necessary to maintain genetic variation with a 95% probability, in both instream and hatchery components.
4	Instream escapement > 50% N_b and hatchery broodstock > 75% N_b.
3	Instream escapement < 50% N_b and hatchery broodstock > 50% N_b.
2	Instream escapement < 50% N_b and hatchery broodstock < 50% N_b.
1	Instream escapement < 50% N_b.

N_b is the effective number of breeders.

C.1.10.5.3 Broodstock Sampling and Mating Protocols

To minimize the genetic and life-history differentiation between hatchery and wild fish, the theoretical ideal for broodstock collection would be to sample the entire breeding population. However, as this is not a practical approach to preserving the wild population, it is generally required that a systematic subsampling scheme be developed that minimizes risk to the wild population. The following broodstock sampling and mating protocols are recommended by NOAA Fisheries (Hard et al. 1992):

- A primary goal of the sampling program should be to obtain a representative sample for use as broodstock while allowing a representative sample to remain in the wild.
- Sampled adults should represent the entire run with regard to size, age, and other measurable phenotypic characters that may have adaptive value.
- If the number of available natural spawners is large enough to permit a large sample to be taken, random sampling (sampling without regard to measurable characters) is likely to ensure that the natural population is represented adequately in the broodstock. If the number of natural spawners is too small to permit a large sample, however, systematic sampling on the basis of measurable characters

(particularly run-timing and size and age at maturity) may be required to achieve adequate representation.

Maintaining genetic characteristics of a population during artificial propagation may also be affected by the manner in which the broodstock are mated. In theory, there would appear to be an advantage to mimic mating strategies that occur in the wild. However, the understanding of patterns of reproductive success in natural populations is incomplete, particularly with respect to males, and NOAA Fisheries consequently discourages attempts to mimic natural spawning behavior in the hatchery (Hard et al. 1992). The mating protocols recommended by NOAA Fisheries consist of the following (Hard et al. 1992):

- The mating design should be chosen to equalize as much as possible the contributions of parents to the next breeding generation. This procedure will maximize N_e for a given number of breeders and minimize the effects of selection.

- If possible, parents should be mated at random with regard to phenotypic characters that may have adaptive value (e.g., age and size at maturity).

- Mating design may include matings of single pairs, matings of single females to overlapping pairs of males, or factorial designs involving crosses between all possible parents. A modified single-pair design is generally preferable to simple matings of single pairs because it reduces risk of loss due to infertile males. A factorial design, assuming that the realized variance in progeny number is small, increases the probability of unique genetic combinations in the progeny. However, a complete factorial design will generally be feasible only with very small populations, since the benefits rapidly decrease (and the logistical difficulties rapidly increase) with increasing numbers of adults.

- Gametes from different individuals should not be mixed prior to fertilization, since mixing would affect the contribution of some individuals if there is variability in the potency of milt.

- In very small populations, a fraction of the milt from each male should be cryopreserved to maintain a "sire bank." These gametes can provide additional male “breeders” in years when the number of available males is low. Moreover, such crosses between brood years can help to preserve long-term genetic variability if severe population bottlenecks have been frequent or persistent.

In summary, broodstock sampling and mating protocols have the potential to affect the wild population primarily through the mechanisms of loss of within-population diversity and outbreeding depression. Table C-64 organizes the potential range of sampling and mating procedures into five categories and provides a score of relative risk level.

Table C-64 Broodstock Sampling and Mating Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	Large, naturally spawning component allowing random mating.
4	Large broodstock with pedigree mating.
3	Large broodstock with random mating; or medium broodstock with pedigree mating.
2	Medium broodstock with random mating; or small broodstock with pedigree mating.
1	Random mating precluded in naturally spawning component (due to small population size and/or isolation).

C.1.10.5.4 Rearing Techniques

Naturalized Rearing Environments

The degree to which artificial selection might be expected to result in the divergence of phenotypes among hatchery-reared adults and or juveniles is related in large part to the difference in selective regimes between the hatchery and natural environment. One obvious method to decrease the potentially deleterious effects of artificial selection is to minimize selective differences between the two environments. One approach for doing so is implementation of the Natural Rearing Enhancement System (NATURES) described by Maynard et al. (1996). The NATURES approach uses naturally colored raceways and rearing ponds, natural substrates, instream cover, subsurface feeding, and lower rearing densities (among other factors) in an effort to mimic natural conditions in the hatchery. Although implementation of these methods may not increase survival *per se*, implementation of NATURES methods might be useful as a means to avoid cryptic side effects of artificial selection. For example, it has been postulated that rearing in environments that have homogeneous flows (e.g., raceways) may result in hatchery-reared progeny that are unable to identify and utilize less energetic low-flow areas in natural streams (Olla et al. 1998). By introducing hatchery-reared progeny to varied currents within a raceway, either by introduction of cover or S-shaped raceways, the NATURES program may increase the ability of hatchery-reared progeny to function effectively under natural conditions.

Environmental conditions in the hatchery that attempt to simulate natural conditions are likely to reduce typical differences between hatchery and natural fish. Examples of naturalized rearing conditions that are being investigated through the NATURES research program include substrate coloration and composition and in-water structures to provide variable water flow conditions. Low rearing density indices (between 0.30 and 0.40 pounds of fish per cubic foot of water per inch of fish length) are recommended by NOAA Fisheries as a means to maximize adult return (Flagg et al. 2000). At a minimum, it is expected that any supplementation or captive brood program would operate under low-density conditions, and that NATURES features would be added as data become more conclusive regarding their benefit to minimizing artificial selection and increasing adult return.

Given that differences in selective regimes experienced by hatchery-reared versus naturally-spawned individuals can contribute to artificial selection, it follows that decreasing the selective gradient between the hatchery and instream environment can minimize the risk of artificial selection. Therefore, it is proposed that one criterion by which the risk of artificial selection can be assessed is the degree to which managers can minimize differences in selective pressures between the instream and hatchery environments. To do so, it is recommended that elements of the NATURES rearing program be implemented, and that the number of NATURES techniques employed be used as the ranking criterion for rearing techniques.

Rearing techniques have potential to affect the wild population primarily through the mechanism of artificial selection. Table C-65 organizes the potential range of rearing techniques into five categories and provides a score of relative risk level.

Table C-65 Rearing Techniques Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	No hatchery captivity.
4	Low-density rearing with multiple NATURES features.
3	Low-density rearing.
2	High-density rearing with NATURES features.
1	High-density rearing.

Fish Health

To minimize the risk of disease transfer to the wild population, hatchery operations should include adequate safeguards for fish health. NOAA Fisheries recommends the following fish health protocols (Hard et al. 1992):

- Adults contributing gametes should be regularly sampled for pathogens of common salmonid diseases.
- Incubation facilities should be sterilized before gametes are transported to them.
- Gametes brought into the facility should be isolated from all others and the resulting fertilized eggs disinfected. To avoid horizontal disease transfer, progeny should be isolated by full-sib family until cleared though pathological testing and then monitored regularly during culture.
- Infected fish should be isolated and treated. However, it should be recognized that some incipient level of disease is natural and also probably essential for immunological readiness for episodic outbreaks.
- If necessary, the hatchery water supply and effluent should be treated to minimize the transfer of pathogens to and from the natural population.

It is assumed that adherence to NOAA Fisheries-recommended guidelines for fish health management will minimize potential risk of disease transfer from the hatchery to wild populations to an undetectable level.

Release Strategies

Age of Releases

Different lifestages of fish may experience differing levels of resource limitation, depending on the time and duration of resource utilization. There is a benefit to fish production goals that minimize temporal overlap in the hatchery-reared and naturally-spawned components, suggesting a preference for smolt release programs over fingerling production. Hatchery-reared fish released as smolts soon migrate to the ocean, and they consequently exhibit little likelihood of competing for freshwater resources utilized by naturally-spawned fingerling rearing within the system. It is recommended that any mitigation or supplementation production program in the Russian River release smolts.

Release Size

The size of a juvenile fish has been shown to affect its ability to compete, escape predators, and survive the ocean phase of its life-history. Stocking with hatchery-reared juveniles of a similar size to naturally-spawned individuals may decrease the probability of competition or predation, and minimize selection pressures that may accompany a clear difference in size. It is recommended that any production programs in the Russian River release smolts within the observed size range of wild smolts.

Acclimation and Volitional Release

If population sizes are small, managers should seek to avoid erosion of between-population diversity by maintaining the highest possible degree of homing fidelity. Straying can be minimized by rearing and acclimation using water from the stream to which adults are hoped to return (Dittman et al. 1994, 1996). Therefore, mitigation for the loss of between-population diversity could be in the form of acclimation facilities.

Fished reared at conservation hatcheries should be released on their own volition, based on the assumption that fish will not leave the hatchery until the physiological process of smoltification triggers their downstream migratory behavior. The time-frame provided for volitional release should mimic the time and age patterns found in wild populations. Within this framework, fish may leave if they wish or remain behind to fend for themselves and smolt, residualize or perish as natural selection takes its course (Flagg et al. 2000). It is important that no attempt be made to reduce natural variation in size at release (Hard et al. 1992).

It is recommended that conservation hatcheries adopt practices aimed at reducing straying to no more than 5 percent. It has been shown that juvenile fish must experience the odors of their natal system at various physiological states to maximize imprinting opportunity. Conservation hatcheries should consequently rear fish for their entire juvenile freshwater lives in water from the intended return location. When this is not possible, a period of acclimation on intended return water should be conducted (Flagg et al. 2000), preferably for a minimum of four weeks.

Release Locations

Releases of fish for supplementation purposes should occur only in locations where the habitat capacity exceeds the requirements of the local naturally spawning population. This indicates the importance for resource managers to identify the area of habitat utilization for various lifestages. Ideally, this information should be used to establish production goals for hatchery operations, specifying production numbers for specific sizes (i.e., lifestages) and release locations. Further, for supplementation-type programs, there is a strong benefit of providing frequent updates to population surveys of both hatchery-reared and naturally-spawned fish. Adaptive management can then be used to evaluate and implement changes in program goals and/or techniques for artificial propagation. For example, the limiting factor in the mainstem Russian River is generally thought to be rearing habitat, particularly for larger juveniles, due to high summer water temperatures (below Cloverdale). Release of steelhead into restored rearing habitat where steelhead have been extirpated or abundance is low, would minimize negative competitive interactions. Monitoring and evaluation over time can provide data to guide future release strategies as steelhead abundance changes.

For the isolated harvest production alternative, managers can consider releasing fish to increase the spatial and temporal separation between hatchery and wild fish. Approaches might include releasing steelhead smolts after wild salmonids have moved out of estuarine habitats, and releasing smolts downstream of habitat used by wild salmonids. Another alternative might consider intentional development of hatchery strains with different migrational timing than other wild salmonids in the basin. However, the potential benefits of this ecological isolation would have to be carefully weighed against the potential risk of developing genetically divergent stocks.

Release strategies have the potential to affect the wild population through several ecological interactions, as outlined previously. Table C-66 organizes the potential range of release strategies into five categories and provides a score of relative risk level. It is assumed that habitat conditions will be surveyed for multiple years prior to juvenile releases to determine appropriate release locations and densities. Although there would be a preference to develop acclimation facilities to allow volitional release in these locations, it is recognized that the large extent of private land ownership in the Russian River watershed is a factor that may affect the opportunity for such facilities.

Table C-66 Release Strategies Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	No hatchery releases.
4	Volitional smolt releases into areas with known habitat carrying capacity.
3	Direct smolt releases into areas with known habitat carrying capacity.
2	Volitional smolt releases into areas with unknown habitat carrying capacity.
1	Direct smolt releases into areas with unknown habitat carrying capacity.

C.1.10.5.5 Duration in Hatchery Captivity

The duration of hatchery captivity has potential to affect the wild population primarily through the mechanism of artificial selection. The rate and extent to which phenotypic, genetic, and behavioral divergence may occur within the hatchery environment is largely dependent on selective pressure within the hatchery, and the number of generations the hatchery-reared stock has been isolated from the donor stock. Typically, divergence requires many generations. However, under intentional selection, there can be evidence of divergence after only two generations (Beasley et al. 1999).

Many sources of artificial selection that could occur in a hatchery can be avoided, such as assuring there is representative sampling of all available broodstock. However, it is not possible to avoid all sources of artificial selection. For example, culling eggs or juveniles exhibiting a high titer for bacterial kidney disease may result in inadvertent selection against those individuals possessing a natural resistance to the disease. All things being equal, one would expect the number of diverged traits and the magnitude of divergence to increase with the duration of captivity. Simply stated, with a longer period of duration, more life-history stages may be subjected to artificial selection, and more traits may become susceptible to the effects of artificial selection. Table C-67 organizes the anticipated range of hatchery captivity into five categories and provides each program alternative with a score of relative risk level.

Table C-67 Duration in Hatchery Captivity Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	No hatchery captivity.
4	Hatchery captivity through fry lifestage.
3	Hatchery captivity through smolt lifestage.
2	Hatchery captivity through adult lifestage.
1	Hatchery captivity for repeated generations.

C.1.10.5.6 Harvest Management

Harvest management has potential to affect the wild population primarily through the mechanism of unintended harvest bycatch of the non-target population. To reduce the potential for deleterious effects, it is essential to monitor the effects of harvest on listed populations. Budget limitations have precluded the ability of CDFG to conduct harvest surveys in recent years, but there is indication that funding may be available soon for such activities (Royce Gunter, CDFG, pers. comm. 2002). Table C-68 organizes the potential range of harvest management decisions into five categories and provides each program alternative with a score of relative risk level.

Table C-68 Harvest Management Evaluation Criteria

Category Score	Evaluation Criteria Categories
5	No harvest allowed within basin.
4	Harvest allowed on one or more non-listed, distinguishable/marked population, with comprehensive surveys to assess harvest, angler effort, and bycatch effects to wild population.
3	Harvest allowed on one or more non-listed, distinguishable/marked population, with moderate survey activity.
2	Harvest allowed on one or more non-listed, distinguishable/marked population, with minimal survey activity.
1	No limits on harvest.

C.1.11 ESTUARY MANAGEMENT

The primary action in the management of the Russian River Estuary (Estuary) is artificial breaching of a sandbar that forms naturally across the mouth of the river. The Estuary goes through a natural cycle of sandbar formation and breaching. When the sandbar closes, it forms a lagoon. When it is open, the Estuary is open to tidal mixing. A sandbar generally forms during the summer and lasts through early fall, and artificial breaching activities would occur during this time. Sandbar formation is primarily influenced by offshore conditions, coastal sand dynamics, and by river flow.

Information on the historical conditions in the Estuary is scarce prior to the construction of Warm Springs Dam and Coyote Valley Dam, the Potter Valley Project, and water management policies under D1610. Before current policies, it is likely sandbar formation occurred much earlier in the year, lasted until ocean conditions or fall rains breached the sandbar naturally, and the Estuary existed as a closed lagoon during the summer (R. Coey, CDFG, pers. comm. 2000). Salmonid migration patterns were likely well-adapted to the natural opening and closing cycles of the Estuary. Formation of the lagoon was also likely to provide excellent rearing habitat for juveniles.

Under D1610, the amount of water that flows to the Estuary during the dry season has resulted in concerns about local flooding and has forced management to breach the sandbar. Therefore the estuarine system no longer supports a lagoon phase.

For baseline flow conditions under D1610, evaluation criteria for water quality were developed assuming an open estuarine system. Under the proposed project, flow to the Estuary would be reduced and eliminating summer breaching of the sandbar is a viable management option.

C.1.11.1 Issues of Concern

Artificial breaching affects water quality in the Estuary, including salinity, temperature, DO, instream cover, and flow rates. Artificial breaching affects salmonid rearing habitat during the summer and fall. Breaching can alter migration patterns in salmonids and potentially flush juveniles out of the lower Estuary before they are ready to go. It may also increase the risk of predation on listed fish species.

The issues related to artificial breaching are summarized as follows:

- Water quality
- Juvenile rearing
- Adult upstream migration
- Juvenile outmigration
- Flushing juveniles out of the Estuary prematurely
- Predation on salmonids

The effects of artificial breaching on salmonid migration and rearing are characterized using the evaluation criteria for water quality. The risk of predation due to breaching is characterized using the evaluation criteria for predation.

C.1.11.2 Water Quality

When the sandbar closes across the river’s mouth, it traps salt water in a lagoon. The denser salt water forms a saltwater lens under the fresh river water (stratification) that traps heat. Through natural processes, DO becomes depleted in the saltwater lens and anoxic conditions can form.

In his studies of central California coast lagoons, Smith (1990) found that the saltwater lens eventually seeps out through the sandbar if it remains closed, and the resulting freshwater conditions provide excellent rearing habitat for steelhead. The rate of conversion to a freshwater system depends on the amount of salt water impounded when the sandbar forms and the amount of river inflow to the system. A sufficient amount of freshwater inflow “freshens” the lagoon and helps to increase the rate of seepage of salt water through the sandbar.

When the sandbar is breached, salt water flows back into the Estuary. Reformation of the sandbar causes salinity stratification to occur and the cycle of freshening starts again. If the sandbar is breached during low-flow periods in the summer, the rate of conversion to a freshwater system can be very slow, and may not occur again that season. This condition results in a return to poor water quality.

If estuaries are managed as open systems, good water quality can be maintained through tidal mixing and/or high river flows (Smith 1990). Such systems require frequent breaching of sandbars to ensure suitable rearing conditions. In a lagoon, good water quality develops when the system is converted to fresh water, which results in lower water temperatures and higher bottom DO levels. Infrequent breaching, especially during low-flow summer months, impairs water quality because salinity stratification results in higher water temperatures and low DO levels (Smith 1990).

Because management of the Estuary under baseline conditions does not allow lagoon formation, the monitoring programs did not address what water quality would be like under a natural estuarine/lagoon cycle.

Under the proposed project, inflow to the Estuary would be reduced during the summer months and managed so that after the sandbar forms, the lagoon would be maintained at a WSE of approximately 8 to 6 feet at the Jenner gage. The lagoon would reach an equilibrium situation in which outflow through the sandbar equals inflow from the river. While deep saltwater pools in the bottom layers might remain, it is likely there would be more surface area and more shallow freshwater habitat, better water quality, and increased productivity, than currently exists under the baseline management scenario. Therefore, the Estuary could be managed as a closed lagoon system.

Potential water quality problems could exist under such a scenario. In November 1992, anoxic water from Willow Creek was flushed into the Estuary when the sandbar was breached at a water level over 9 feet (RREITF 1994). This could have been caused by a flush of anoxic sediments. Alternatively, poor water quality could have formed when vegetation was submerged by high water levels and began to decay. However, with stable WSE, aquatic vegetation could establish and summer water quality in Willow Creek could potentially improve. Finally, agricultural runoff and treated sewage discharge may increase nutrient levels in the Estuary. Reduced river flow in the lower river may reduce dilution of these nutrients.

Part of Merritt Smith Consultants' (MSC) and SCWA's 5-year monitoring effort has focused on Willow Creek. In 1992, a fish and invertebrate kill was associated with a flush of anoxic water from Willow Creek after the sandbar was breached when water levels were over nine feet. At high water levels, larger areas of the marsh in Willow Creek are inundated, and a large water volume may have become anoxic. This kind of event has not occurred during 5 years of monitoring in the Estuary (MSC 2000). Mortality of prickly sculpin in 1998, associated with a breaching event after water levels rose to 8.2 feet, may have been caused by low DO in water draining from Willow Creek, but no anoxia was detected (MSC 1998). Dead dungeness crabs were found in 1999 near the mouth of Willow Creek, but this was most likely due to a flush of fresh water after an artificial breaching event (MSC 2000). Artificial breaching of the sandbar is currently conducted at lower water elevations on the Jenner gage. Breaching below approximately 7 feet at Jenner appeared to prevent the outflow of anoxic water from the creek.

C.1.11.3 Evaluation Criteria for Water Quality

Infrequent artificial breaching during the dry season would impair water quality. Optimal water quality conditions would result if the sandbar were to remain closed and the lagoon were allowed to convert to freshwater conditions. If inflow to the lagoon were high enough to cause flooding of local property and an artificial breaching were needed, frequent breaching would be needed to limit the duration and intensity of poor water quality events.

C.1.11.3.1 Sandbar Open

Biological and water quality monitoring in the Estuary over a 5-year period shows that water quality begins to deteriorate immediately after the sandbar forms. When the sandbar opens, tidal flushing can result in improved water quality. Evaluation criteria for water quality under an open sandbar management strategy are based on the assumption that limiting the amount of time the sandbar remains closed following a closure event would limit the severity and length of poor water quality events (Table C-69). A score of 5 is given to a breaching schedule that ensures the sandbar is not closed for more than 5 days. Sandbar closed episodes during the monitoring study occasionally exceeded 10 days, and it is estimated that in general, closure of the sandbar for longer than 14 days may result in water quality conditions that are detrimental for salmonids. Longer periods of time are given lower scores.

Table C-69 Water Quality Evaluation Criteria — Sandbar Open

Category Score	Frequency of Artificial Breaching (Time Sandbar Remains Closed)
5	0 – 5 days
4	6 – 10 days
3	11 – 14 days
2	15 – 21 days
1	> 22 days < 40 days

C.1.11.4 Rearing and Migration

Estuaries and lagoons provide important rearing habitat for salmonids. Smaller lagoons in the Central California Coast Steelhead ESU and small lagoons north of the Russian River have been shown to provide important rearing habitat for steelhead in the summer (Smith 1990, Larson 1987, Anderson 1995, 1998, 1999, Cannata 1998). Because they are food-rich, lagoons can contribute substantially to juvenile growth, which can translate into increased return rates for adults (Smith 1990).

Lower river environments in the north (most of which are estuaries open to tidal mixing) also provide important habitat for Chinook salmon fry or fingerling rearing (Reimers 1973, Levy and Northcote 1982, Kjelson et. al 1982, Simenstad 1982, Anderson and Brown 1982, Meyers and Horton 1982, Groot and Margolis 1991). Reimers (1973) demonstrated that Chinook salmon exhibiting a life-history strategy that remained in fresh water until early summer, then reared for a period of improved growth in the Estuary, represented approximately 90 percent of the returning spawners in the Sixes River, Oregon. In the Sacramento-San Joaquin River estuary, Chinook salmon fry rear in freshwater habitat within the upper delta before moving into the estuary as smolts (Kjelson et al. 1982).

Juvenile salmonid rearing is generally thought to occur during the following times:

- Coho Salmon — All year, generally rear in tributaries
- Steelhead — All year
- Chinook Salmon — February through June

Salmonid survival necessitates their ability to pass through the river mouth and Estuary during migration periods, and that water quality is high during passage. Artificial breaching provides more passage opportunities than would occur under natural conditions. A key consideration is whether water quality is sufficient when additional passage occurs.

When the rainy season begins, the sandbar generally opens naturally. Rain and increased flow at this time would create good passage conditions for adults migrating upstream. The peak adult Chinook salmon spawning run begins after November, although Chinook salmon begin to gather outside of the river in mid-August.

Adult migration periods for salmonids are:

- Coho Salmon — November through January
- Steelhead — January through March
- Chinook Salmon — Mid-August through January, with peaks occurring after November

Water quality in the Estuary is primarily dependent upon how long the sandbar is closed, and sandbar closure is primarily related to ocean and river flow conditions. Once the sandbar is breached, water quality does not immediately improve in the upstream parts of the Estuary, and the sandbar may close again before it does improve. Several successive breaching events may be required to improve water quality in upper reaches.

If the sandbar were to be breached before winter storms help improve water quality in the mainstem Russian River, adult Chinook salmon may not be able to pass, and may become trapped in poor quality water. Coho salmon and steelhead adults generally migrate later, and are more likely to move upstream when water quality has improved with higher flows.

Water quality evaluation criteria under Estuary management are applied for adult salmonid passage from August to the first significant rains. If the rains are very late, artificial breaching may provide passage during peak spawning times while water conditions could still be poor in the Estuary or the mainstem river. Anecdotal information may provide information on whether salmonids, particularly Chinook salmon, have been trapped anywhere in the river.

Smolt emigration is usually complete by early summer. If the sandbar were to close at some time in the late spring, artificial breaching would provide additional passage opportunities in addition to those that would have naturally occurred. This may benefit salmonids that have undergone the physiological changes that prepare them for saltwater conditions. Juvenile salmonid migration generally correlates to the occurrence of spring freshets, among other factors, and water quality at this time would be expected to be better than during the summer in the Estuary and the river. Emigration times for juveniles are:

- Coho Salmon February through mid-May
- Steelhead March through June
- Chinook Salmon February through May

Water quality evaluation criteria under sandbar-open management are applied for juvenile migration in the spring (Table C-69). Furthermore, artificial breaching affects the amount of time a closed sandbar could delay juvenile outmigration.

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ATTACHMENT 1

WATER TEMPERATURE CRITERIA AND REFERENCES FOR SALMONIDS

Table A Water Temperature Criteria and References for Salmonid Upstream Migration

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Coho Salmon	Upstream Migration	0	1.7	< 3	ILT	Bjorn and Reiser
		1	3	< 4		
		2	4	< 5		
		3	5	< 6		
		4	6	< 7.2		after McMahon 1983
		5	7.2	12.7	Migration Range	Bjornn and Reiser, Wedemeyer 1970
		4	12.7	14		Burroughs, Bjornn and Reiser
		3	14	< 15		after McMahon 1983
		2	15	< 16		after McMahon 1983
		1	16	< 21.1		after McMahon 1983 with Hallock (1970) as an estimator
		0	21.1			Use Hallock as an estimator
Steelhead	Upstream Migration	0	4		No Migration	cited by Raleigh 1984
		1	4	5		
		2	5	6		
		3	6	7		
		4	7	7.8		
		5	7.8	11	Migration Range	CDFG 1991
		4	11	13		
		3	13	15		
		2	15	17		
		1	17	19		
		0	21.1		No Migration	Cramer 1992

Table A Water Temperature Criteria and References for Salmonid Upstream Migration (Continued)

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Chinook Salmon	Upstream Migration	0	<0.8		ILT	Bjornn and Reiser
		1	□3	> 0.8		
		2	≤ 5.2	>3		
		3	≤ 7.9	> 5.2		
		4	≤ 10.6	> 7.9		
		5	> 10.6	≤ 15.6	Migration Range	Bjornn and Reiser
		4	> 15.6	≤ 17.0		
		3	> 17.0	≤ 18.4		
		2	> 18.4	≤ 19.8		
		1	> 19.8	≤ 21.1		
		0	>21.1		No Migration	Hallock 1970, Cramer 1992

Table B Water Temperature Criteria and References for Salmonid Spawning

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Coho Salmon	Spawning	0	1.7		ILT	Bjornn and Reiser
		1	1.7	3		McMahon 1983
		2	3	4		
		3	4	6		
		4	6	7		McMahon 1983
		5	7	13	Columbia	EPA 1974
		4	13	14		McMahon 1983
		3	14	15		
		2	15	16		
		1	16	17		
		0	17			McMahon 1983
Steelhead	Spawning	0		4	No migration criteria	Hanel (1971) McMahon 1983
		1	4	5		
		2	5	6		
		3	6	7		
		4	7	7.8		
		5	7.8	11.1	American River	CDFG 1991 Rich 1987
		4	11.1	14	embryos	Raleigh et al. 1984
		3	14	16		
		2	16	18		
		1	18	20		
		0	20		unsuitable	Raleigh et al. 1984

Table B Water Temperature Criteria and References for Salmonid Spawning (Continued)

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Chinook Salmon	Spawning	0		1	embryo survival	Seymour(1956)
		1	1	2.5		
		2	2.5	3.5		
		3	3.5	4.5		
		4	4.5	5.6		McMahon 1983
		5	5.6	13.9		Bjornn and Reiser Bell (1986)
		4	13.9	14.5		
		3	14.5	15.2		
		2	15.2	16		
		1	16	16.7		
		0	16.7		embryo survival	Boles 1988

Table C Water Temperature Criteria and References for Salmonid Incubation (Continued)

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Coho Salmon	Incubation	0		0		McMahon 1983
		1	0	3		McMahon 1983
		2	3	3.5		McMahon 1983
		3	3.5	4		McMahon 1983
		4	4	4.4		McMahon 1983
		5	4.4	13.3		Bjornn and Reiser Bell (1986)
		4	13.3	14		McMahon 1983
		3	14	15		McMahon 1983
		2	15	16		McMahon 1983
		1	16	18		McMahon 1983
		0	18			McMahon 1983
Steelhead	Incubation	0		1.5	unsuitable	Raleigh et al. 1984
		1	1.5	3		
		2	3	4.5		
		3	4.5	6		
		4	6	7.8		
		5	7.8	11.1	American River	CDFG 1991 Rich 1987
		4	11.1	13		
		3	13	15		
		2	15	17		
		1	17	20		Raleigh et al. 1984
		0	20		unsuitable	Raleigh et al. 1984

Table C Water Temperature Criteria and References for Salmonid Incubation

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Chinook Salmon	Incubation	0		1	No survival	Raleigh et al. 1986 Seymour (1956)
		1	1	2		
		2	2	3		
		3	3	4		
		4	4	5		
		5	5	12.8		Boles et al. (1988) Seymour (1956)
		4	12.8	14.2	Sacramento River, 10 percent Mortality	Resources Agency (1989)
		3	14.2	15		
		2	15	15.8		
		1	15.8	16.7		
		0	> 16.7		Sacramento River, 100 percent mortality	Resources Agency (1989) Boles(1988)

Table D Water Temperature Criteria and References for Salmonid Rearing

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Coho Salmon	Rearing	0	1.7		ILT	Bjornn and Reiser Brett (1952)
		1	1.7	4		
		2	4	7		
		3	7	8		
		4	8	12		McMahon 1983
		5	12	14	preferred	Bjornn and Reiser Brett (1952)
		4	14	15		McMahon 1983
		3	15	16		McMahon 1983
		2	16	20		McMahon 1983
		1	20	26		McMahon 1983
		0	26		ILT	Bjornn and Reiser Brett (1952)
Steelhead	Rearing	0	0		ILT	Bjornn and Reiser Bell (1986)
		1	0	2		Raleigh 1984
		2	2	4		
		3	4	8		
		4	8	12.8		
		5	12.8	15.6	American River Rearing	CDFG 1991 Rich 1987
		4	15.6	18		
		3	18	20		
		2	20	22		
		1	22	23.9		
		0	23.9		ILT	Bjornn and Reiser Bell (1986)

Table D Water Temperature Criteria and References for Salmonid Rearing (Continued)

Species	Lifestage	Category	Temperature (°C)		Criteria	References
Chinook Salmon	Rearing	0	1		ILT	Bjornn and Reiser Brett (1952)
		1	1	4		
		2	4	6		
		3	6	8		
		4	8	12		
		5	12	14	preferred	Bjornn and Reiser Brett (1952)
		4	14	17		
		3	17	20		
		2	20	23		
		1	23	26		Raleigh 1986
		0	26		ILT	Bjornn and Reiser Brett (1952)